A Spatio- Temporal Ammonia Emissions Inventory for the UK

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Declaration

I, Sofie Hellsten, hereby declare that this thesis has been composed solely by myself and that the work presented herein is my own unless otherwise stated. This thesis has not been previously presented or submitted for any other degree.

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Abstract

Ammonia (NH_3) emissions to the atmosphere have increased during the last century, mainly due to the intensification of agricultural production, which is the main source of NH_3 emissions. Ammonia is regarded as a pollutant responsible for many detrimental effects in the environment, including eutrophication, acidification and species composition change. Spatial NH_3 emission inventories are used as input to atmospheric transport models and hence are a valuable source of information when assessing the environmental impact of NH_3 emissions. The aim of the present study is to provide a new, updated spatial NH_3 emission inventory for the UK, building on the established AENEID model (Atmospheric Emissions for National Environmental Impacts Determination). A major objective in this study has been to assess the uncertainties in, and to improve the modelling results of, the AENEID model. Improvements to the model were implemented for the distribution methodology of pig and poultry manure, a spatially varying cattle grazing season was incorporated, and the temporal resolution improved from annual to monthly emission estimates. A variable emission source strength was incorporated by linking the model with the NARSES N-flow model, a current tool used in the UK for assessing abatement efficiencies. Uncertainties in the spatial NH_3 emission inventory resulting from the aggregation of zones in the agricultural statistics were also assessed, showing that such aggregation has a significant impact on the estimated location of emissions.

The new distribution methodology for emissions from pig and poultry manure is based on an iterative process, taking into account the "manure saturation rate" to agricultural land, and provides more realistic results than the original model where all emissions were distributed within the parish of origin. The cattle grazing season in the UK varies significantly throughout the country and also between years. By incorporating a sub-model into AENEID, it was shown how the emission results would be affected, due to the variation in the proportions of housing emissions, associated storage and spreading emissions and grazing emissions. The monthly ammonia emission maps showed a strong seasonal emission pattern, with the highest modelled emissions occurring in springtime and the lowest emissions during summer.

The total UK agricultural NH_3 emission for year 2000, based on the new AENEID approach, was estimated at 221 kt NH_3-N or 268 kt NH_3. A comparison of the model results with previous inventories has shown that the overall emission estimate has decreased, and unrealistically high and localised emission peaks have been reduced.

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1 Background - The problem of ammonia emissions and the need for spatial emission inventories

1.1 Introduction

Ammonia emissions to the atmosphere have increased rapidly during the last century. It has been estimated that the emissions have doubled since 1950 (Asman et al., 1988, 1998), and the consequences of this increased emission is now becoming evident with species composition change, eutrophication and acidification. Therefore action needs to be taken to decrease the emissions to reduce the risk of further environmental damage.

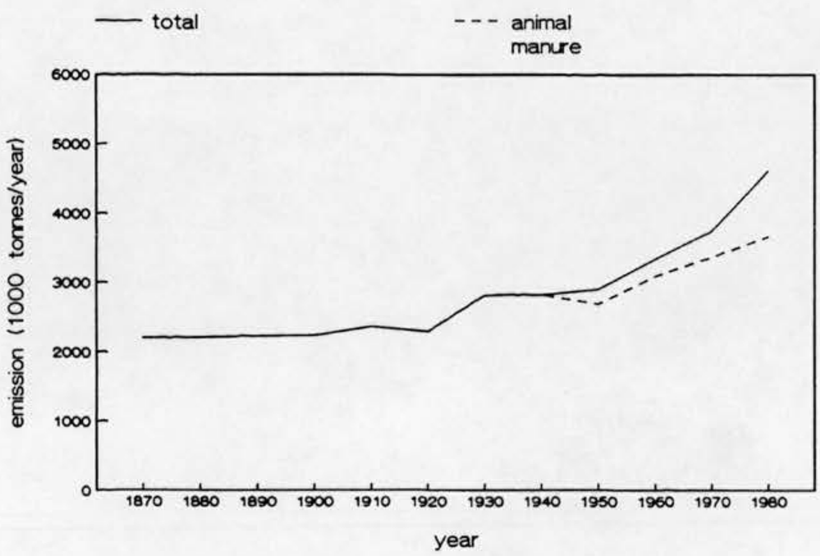

Figure 1.1. Ammonia emissions from agriculture in Europe, 1870 - 1980, as estimated by Asman et. al (1988).

Moriarty (1990) defines a pollutant as *'a substance that occurs in the environment, at least in part, as a result of human activities, and which has a deleterious effect on the environment'*.

Thus, according to Moriarty's definition, a natural component, derived from natural processes may also be considered a pollutant, if present at unexpectedly high concentrations. "Natural pollutants" can in fact have just as serious environmental consequences as the more conventional anthropogenic emissions, such as heavy metals and acidifying species.

Ammonia (NH_3) emissions derive from both natural and anthropogenic sources and processes. The main source is agriculture, with the largest part originating from natural processes, i.e. excretion of urine and dung. Although these processes are considered 'natural', the emissions they generate become anthropogenic in nature, due to the concentrating effect of human activities and the fact that they occur in 'un-natural' amounts.

A century ago ammonia would probably not have been considered a pollutant according to Moriarty's definition, since it mainly occurred in small quantities. Today the intensification of agriculture with more livestock, higher stocking rates, and industrial scale pig and poultry production etc, has led to a dramatic increase in ammonia emissions to the atmosphere. In addition, anthropogenic sources of ammonia have also increased rapidly, e.g. industrial fixation of nitrogen for use as fertilizer (see Figure 1.2). In 1912, the first factory to produce synthetic ammonia through the Haber-Bosch process, a chemical process to convert unreactive N_2 to NH_3, started up (Manchester, 2002). This process enables an unlimited supply of nitrogen to be provided to the fertilizer industry, and today the Haber-Bosch process is used to produce nitrogen fertilizers to support more than half of the food production in the world (Galloway and Cowling, 2002).

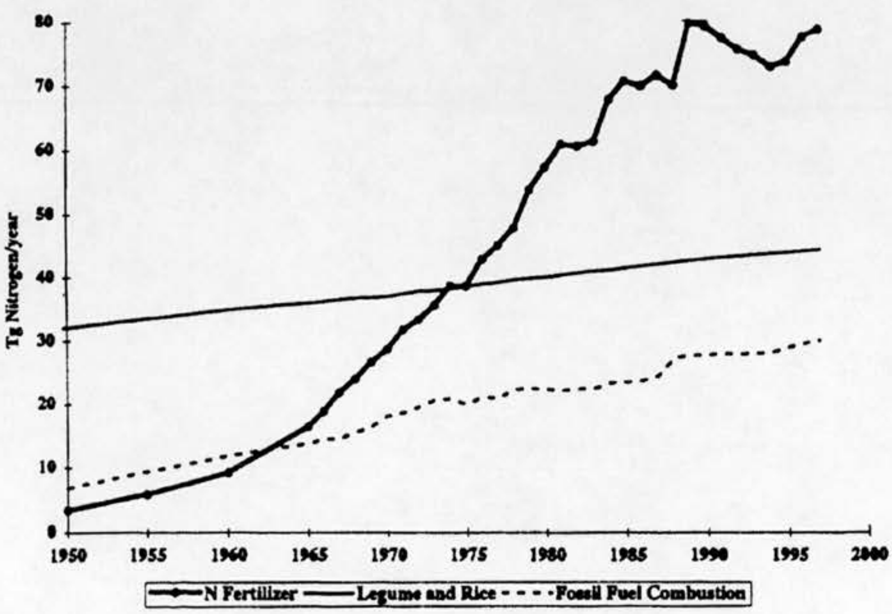

Figure 1.2. Temporal trends (1950 – 1997) in the antropogenic production of reactive N from fertilizer production, fossil fuel combustion and nitrogen fixation from legumes and rice. Source: Galloway (1998).

As a consequence of the intensification of agriculture and the increased levels of anthropogenic ammonia emissions, signs of detrimental effects are evident in the environment (e.g. Van der Eerden et al., 1991; van der Meer, 2001). These effects are occurring in many sensitive semi-natural ecosystems, mainly near agricultural areas. Studies have shown that small nature reserves and edges of semi-natural areas in agricultural landscapes are at particular risk of high NH_3 deposition rates (Sutton *et al.*, 1998; Dragosits *et al.*, 2002), which justifies fine scale assessments of the effect of ammonia emissions in agricultural landscapes.

1.2 Sources and fate of ammonia emissions

The main source of ammonia is agriculture (85 – 90 %), mainly due to the volatilisation of NH_3 from livestock manures. About 248 kt NH_3 of ammonia are released from agriculture every year in the UK (Misselbrook *et al.*, 2004), and the remaining 10 - 15 % (about 40 kt) are derived from non-agricultural sources (Dragosits and Sutton, 2003; Dragosits *et al.*, 2004).

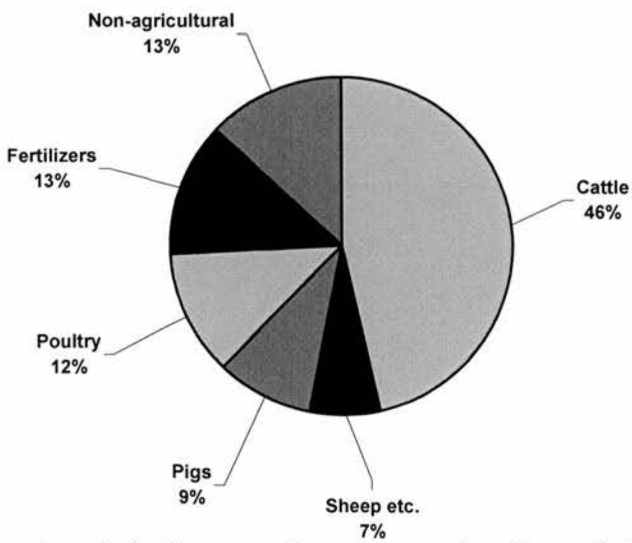

Figure 1.3. UK ammonia emission by source. Percentages are based on emission calculations for 2002 (Misselbrook *et al.*, 2003; Dragosits *et al.*, 2004).

1.2.1 Agricultural sources

The main agricultural sources of NH_3 are management of livestock manure and nitrogen fertilizers. Livestock only utilise a small amount of the nitrogen contained in their feed, and the excess N is excreted as dung and urine (manure). When the excreta get in contact with air, ammonia easily volatilises (Monteny and Erisman, 1998). Ammonia emissions from livestock manure are usually associated with four animal husbandry stages: grazing, housing of livestock, manure storage and land spreading of manure. Manure spreading on land contributes the largest proportion of ammonia emission from manure management in the UK, closely followed by housing emissions from livestock buildings, with grazing emissions and storage emissions as the smaller components (Misselbrook *et al.*, 2003). In Chapter 2, aspects of UK agriculture relevant to ammonia emissions are outlined and discussed in detail.

1.2.2 Non-agricultural sources

Non-agricultural sources of ammonia have been neglected in many emission inventories in the past. Although most individual non-agricultural emission sources only contribute small amounts, together they make up about 10-15 % of the total UK ammonia emission. Non-agricultural sources include humans, pets, wild animals, industrial sources, biomass burning, sewage, landfill and transport (Sutton *et al.*, 2000a). Non-agricultural sources are further discussed in Chapter 3.

1.2.3 Fate of ammonia emissions

Once ammonia has been released into the atmosphere, it is difficult to predict its fate, which is affected by many factors such as type of source, meteorological conditions (wind speed and direction, temperature and humidity), turbulence, atmospheric stability and, indirectly, topographic effects which affect the meteorology (Kiely, 1997). Once emitted, NH_3 may be transformed to other species (e.g. NH_4^+), undergo atmospheric transport and finally be deposited (Aneja *et al.*, 2001), (see Figure 1.4).

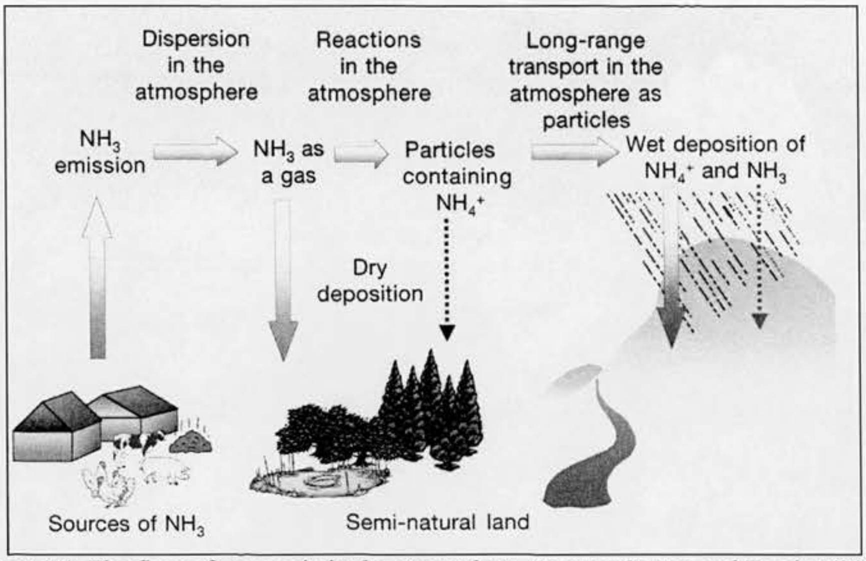

Figure 1.4. The flow of ammonia in the atmosphere. Source: Sutton and Fowler (2002).

Transformation

When ammonia is emitted to the atmosphere, it easily reacts with acidic aerosols, e.g. sulphuric acid and gaseous nitric acid, forming ammonium sulphates and nitrates (aerosols) (Singles *et al.*, 1998). This reaction is strongly driven by ammonia being an important base compound in the atmosphere that neutralises acids and transforms into NH_4^+ (the ammonium ion). Ammonia (NH_3) and ammonium (NH_4^+) are generally referred to as 'reduced nitrogen' (NH_x) (Sutton and Fowler, 2002).

Transport

Both ammonia and ammonium can be transported in the atmosphere. The transport distance is strongly dependent on the reaction rate, because ammonia (NH_3) and ammonium (NH_4^+) have different lifetimes in the atmosphere. The reaction rate in turn depends on factors such as the height above ground level, humidity, temperature and acid concentration (Asman *et al.*, 1998). Ammonia (NH_3) tends to have a shorter transportation distance than NH_4^+ due to its reactive nature, and therefore deposits close to the source (Sutton *et al.*, 1998). Ammonium (NH_4^+) on the other hand can remain airborne longer and can therefore be transported long distances, before being deposited (Sutton and Fowler, 2002).

Deposition

Ammonia can be deposited in two different ways, either by dry deposition of NH_3 or as wet deposition of ammonium (NH_4^+). In the process of dry deposition, gases and particles are directly transferred to surfaces such as soil and plants, while wet deposition is a result of ammonium being dissolved in water droplets and deposited in rainfall, snow or hail (Brimblecombe, 1996).

Dry deposition mostly occurs close to the source (within a few kilometres) and shows a high spatial variability, with NH_3 concentrations decreasing rapidly with distance from the source (Asman *et al.*, 1998; Sutton *et al.*, 1998). Ross and Jarvis (2001) showed that between 20 and 60 % of the NH_3 emitted from urine was deposited within 2 m of a source (an artificial urine patch), although results of Theobald *et al.* (2004b) suggest that this may be an overestimate. Studies have shown that important variables influencing dry deposition include source height, wind speed, atmospheric stability, surface concentration, surface resistance, surface roughness length and compensation points (see section 2.3.6) (Sutton *et al.*, 1993; Asman, 1998; Sutton *et al.*, 2001a; Theobald *et al.*, 2004b). Wet deposition is more important at distances further from the source and is the dominant deposition mechanism at remote locations far away from sources (Asman *et al.*, 1998).

1.2.4 The wider N-cycle

It is important to highlight that not only ammonia emissions are responsible for nitrogen deposition, as NO_x and its reaction products (HNO_3 and NO_3^-) also contribute to nitrogen deposition (Asman *et al.*, 1998). Nitrogen exists in various forms (e.g. N_2, NH_3, N_2O, NO_x, NO_3^-) and moves between different mediums (atmosphere, water, soil, plants, animals and humans), as shown in Figure 1.5. Most N exists in its stable form as N_2 in the atmosphere, but a single N atom can have cascading impacts, since it can be converted to any other N species in favourable conditions (Galloway, 1998). Table 1.1 summarises the most important processes transforming N in the global nitrogen cycle, including nitrification, denitrification, nitrogen fixation, mineralisation, immobilisation and volatilisation (Kiely, 1997). In addition, gaseous N can be converted into inorganic N through either the natural process of lightning, or through artificial energy-intensive industrial processes such as the Haber-Bosch process. The most important process in the context of ammonia emissions is volatilisation, which is further discussed in Chapter 2.

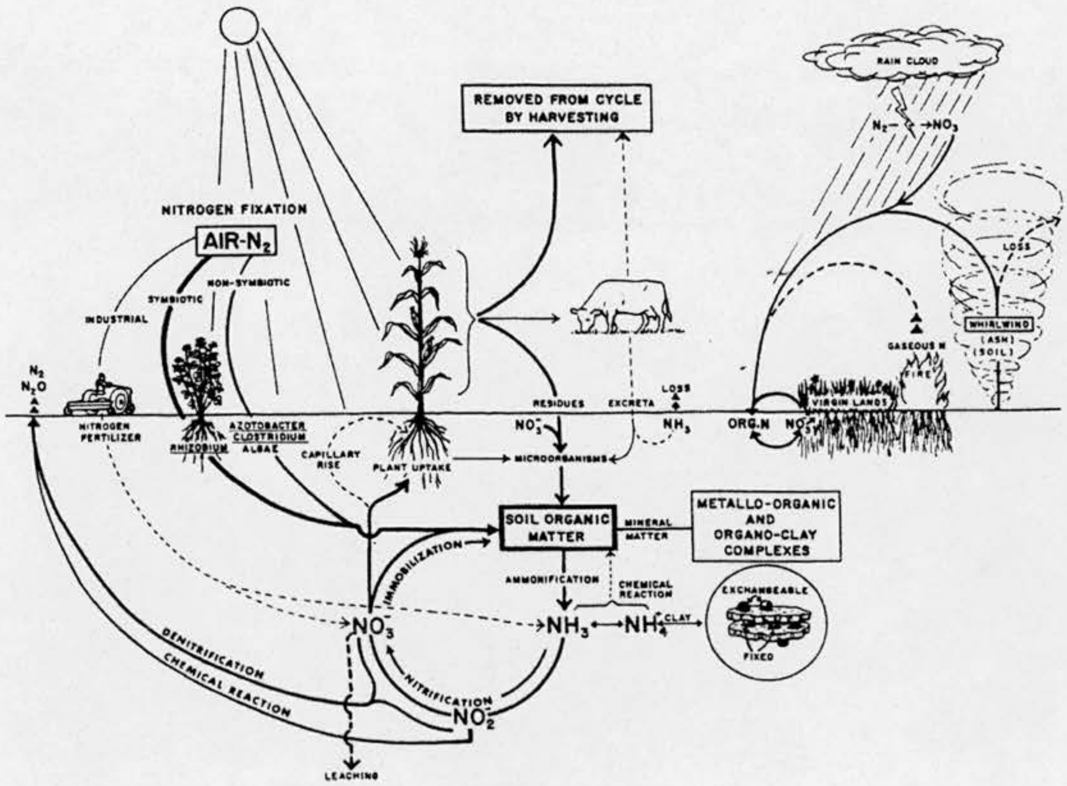

Figure 1.5 The nitrogen cycle. Source: Stevenson (1986).

Table 1.1. Important processes transforming N in the nitrogen cycle.

Nitrification	Nitrifying bacteria convert NH_x to NO_3^-.
Denitrification	Transformation of NO_3^- into gaseous N by heterotrophic bacteria.
Nitrogen fixation	N_2 is fixed from the atmosphere by nitrogen fixing crops or bacteria.
Mineralisation	Organic N is oxidized or transformed to inorganic forms (e.g. NH_4^+), thereby made available to plants.
Immobilisation	Inorganic N is transformed into organic form by soil micro-organisms or plants.
Volatilisation	Transformation of NH_4^+ into NH_3 that is released into the atmosphere.

When reactive N in any form is accumulating in pools in the environment, either in the atmosphere, the soil or in water, the 'natural' nutrient balance is disturbed and this can cause detrimental environmental effects. Reactive nitrogen will accumulate in an ecosystem, transfer to another system, or undergo denitrification into N_2. However, denitrification cannot always keep up with the rate at which reactive nitrogen is created and deposited. N is therefore accumulating in the environment as the inputs and outputs of reactive N are not in balance (Galloway and Cowling, 2002). It is even likely that global anthropogenic N inputs will increase even more in the future due to increased global population and increased animal protein in human diets. Animals consume N, but the excess N not incorporated into tissues or milk is excreted as waste in both organic and inorganic forms, and this excess nitrogen can leak to the environment either as NH_3, N_2O or NO_3^- (Kiely, 1997). Galloway and Cowling (2002) suggest that only 14 % of the nitrogen in the production cycle of plant protein food ends up in humans, with the remaining 86 % being either recycled within the agricultural system, or lost to the environment. They predict that the efficiency of growing animal protein for humans is even worse, with as much as 96 % recycled or lost to the environment (Figure 1.6). A large proportion of the nitrogen lost in the agricultural system is lost in the form of ammonia emissions.

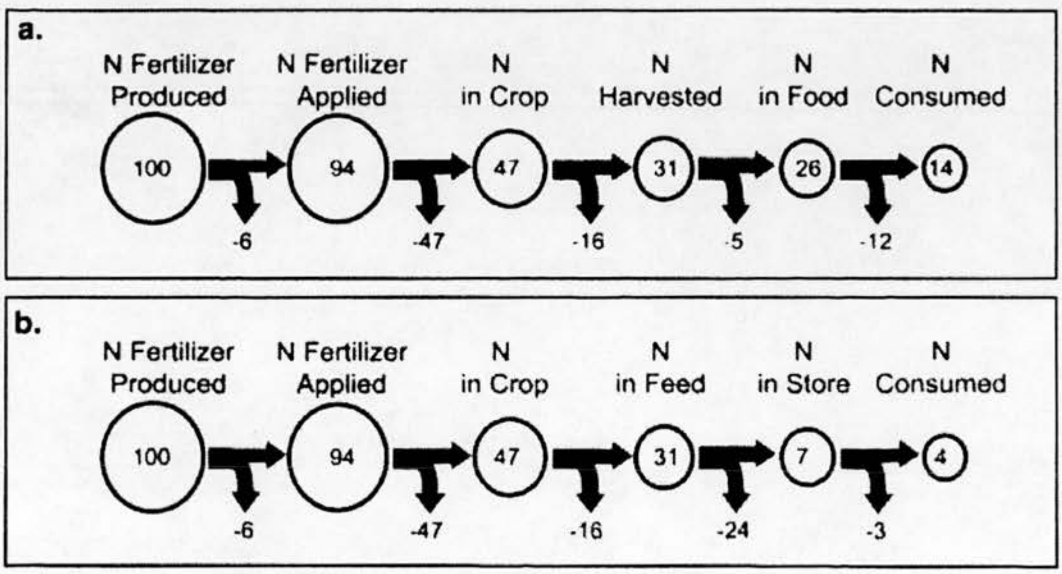

Figure 1.6. The fate of fertilizer N produced by the Harber-Bosch process from factory production to human consumption for a) vegetarian diet, and b) carnivorous diet. Source: Galloway and Cowling (2002).

1.3 Environmental effects

Although intensification has led to increased yields in agriculture, it has been at the cost of environmental damage, such as increased NH_3 emissions. The direct effects of increased concentrations of ammonia are less of a problem than the indirect effects of nitrogen deposition caused by ammonia emissions (Sutton and Fowler, 2002). Effects due to increased nitrogen deposition include (see Figure 1.7):

- Increased nitrate and acidity concentrations in soil, ground water and surface waters
- Change in species composition
- Loss of species of conservation importance
- Damage to vegetation
- Health effects
- Climatic change
- Changes to the biological processes in the soil

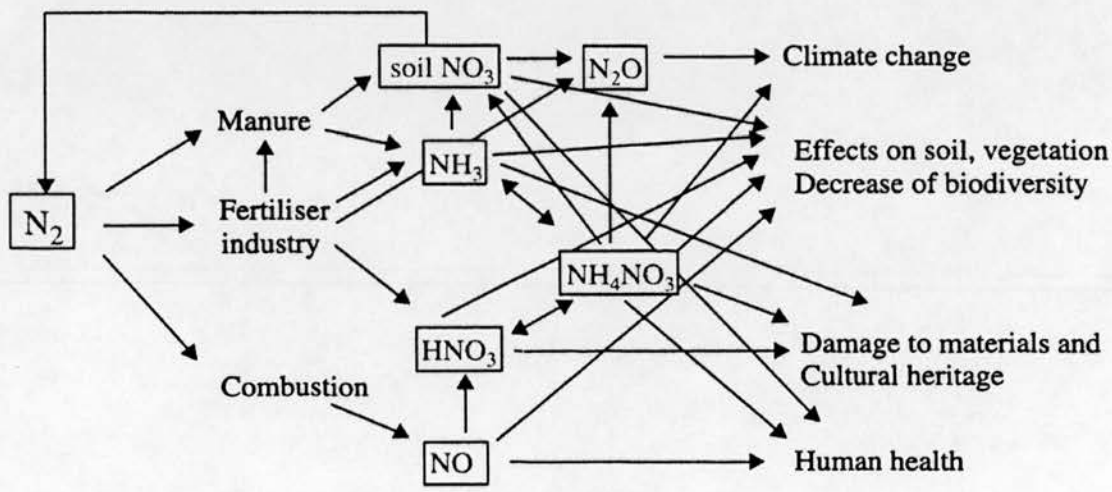

Figure 1.7. Sources and effects of nitrogen in the environment. Source: Erisman *et al.* (2003).

The problem with ammonia emissions is that it may take a long time before damage to the environment is noticeable, because of the accumulation of N in various reservoirs, e.g. due to the buffering capacity of many soils (NEGTAP, 2001; Galloway and Cowling, 2002). At present it may therefore be difficult to estimate the full future impact on the environment from today's emissions.

1.3.1 Eutrophication

The most severe and long lasting effect of emissions of ammonia to the environment is eutrophication, i.e. nutrient enrichment in ecosystems. The impact of eutrophication depends on the type of ecosystem. Crops and fertilized grassland are not adversely affected by N deposition as they already receive large doses of N fertilizers, whereas sensitive habitats adapted to a low N supply e.g. heaths, moors, blanket bogs and semi-natural grasslands are at risk of changes to the species composition (Sutton *et al.*, 1993; NEGTAP, 2001). When reduced nitrogen is deposited in the environment, N is increased in the ecosystem if denitrification occurring in the soil cannot cope with the additional N input (Brimblecombe, 1996). This build-up of nitrogen can lead to changes in plant composition as nutrient sensitive plants are replaced by more competitive species which thrive on the additional N supply (Sutton *et al.*, 1993). This loss of sensitive species in a historically balanced ecosystem is likely to have an adverse 'knock-on' effect on other species within that ecosystem, especially if a 'key' species is lost (Kiely, 1997). This 'knock-on' effect and the complexity of ecosystems make it difficult to predict the full impact and the overall reduction in biodiversity as a result of N enrichment. Aquatic ecosystems are also affected by eutrophication, e.g. algal growth may increase, and the oxygen balance may be disturbed, with a resulting loss of fish and deterioration in water quality (Convery and Roberts, 2000).

1.3.2 Acidification

As with eutrophication, acidification can also lead to changes in species composition and results in acid-sensitive species being lost in favour of more acid resistant species (Hornung *et al.*, 2002). This increased acidity induces chemical changes in soil, making toxic ions like Al^{3+} come into solution, which can harm micro organisms, plants and animals (Krupa, 2003). Other elements that are vital to plants, such as K^{+}, Ca^{2+} and Mg^{2+}, may become unavailable, disturbing the nutrient balance in the soil.

Acidification (see Figure 1.8) takes place in the soil if the ammonium N is oxidised to nitrate by bacteria (Equation 1.1) or absorbed by soil organic matter or biomass (Equation 1.2) instead of being used for plant growth (Sutton *et al.*, 1993).

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+ \quad (1.1)$$

$$NH_4^+ \rightarrow R\text{-}NH_2 + H^+ \quad (1.2)$$

R is the biomass or soil organic matter

Acidification is not solely due to ammonia deposition, but also due to deposition of nitrogen oxides and sulphur dioxide. Historically, sulphur deposition has been the main cause of acidification, but as emissions of sulphur oxides have been reduced drastically during the past decades, the importance of reduced nitrogen for acidification has increased relative to other acidifying pollutants (NEGTAP, 2001).

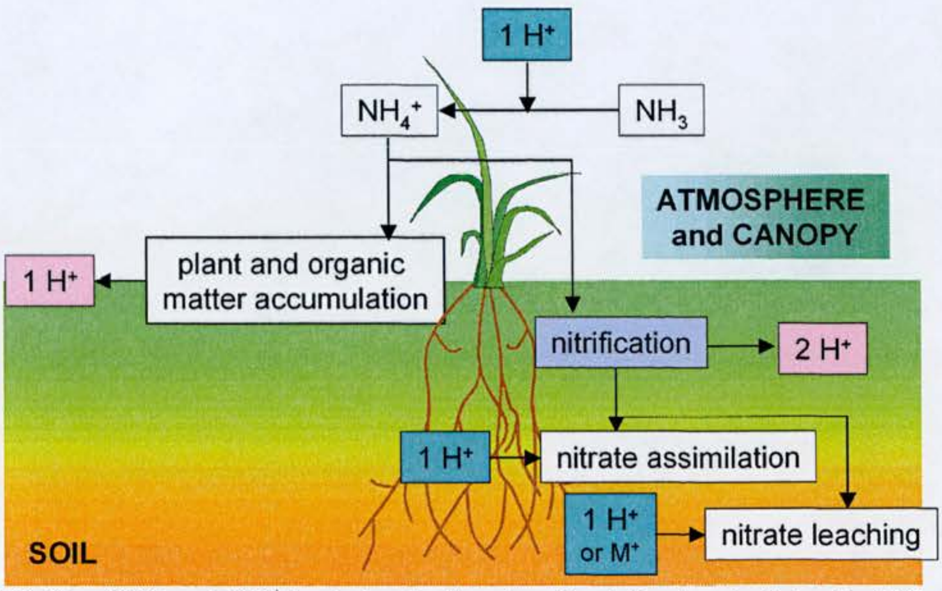

Figure 1.8. Fate of NH_3 and NH_4^+ in plant and soil systems in relation to potential soil acidification (from NEGTAP, 2001, after Sutton *et al.* 1993).

1.3.3 Aerosol formation and climatic change

Secondary particulate matter (PM), i.e. aerosols, has been shown to cause many detrimental health effects and deteriorating visibility in the atmosphere due to haze (Schaap *et al.*, 2002). Another negative impact of aerosols is that they contribute to climatic change, because of the cooling effect occurring when aerosol particles reflect the solar radiation and scatter it back into space (Ten Brink *et al.*, 1997).

Aerosols are created by reactions involving many chemical species, e.g. SO_2, NO_x and NH_3 (Erisman and Schaap, 2004). Ammonia is an important element in aerosol formation in the atmosphere, because it easily reacts with sulphuric and nitric acids due to its alkaline nature, forming ammonium nitrate and ammonium sulphate aerosols. Recent research has shown that the importance of ammonia for aerosol formation has increased relative to SO_2 (Erisman and Schaap, 2004). As SO_2 emissions decrease, nitrate can compensate for the decline in sulphate. Reducing the formation of secondary particulate matter (PM) can therefore only be effective if ammonia is reduced along with SO_2 and NO_x.

Apart from aerosol formation, another indirect climatic impact of ammonia emissions is the release of N_2O from N saturated soils when nitrogen undergoes denitrification, resulting in N_2 or N_2O (Krupa, 2003). N_2O is a climate relevant gas that is contributing to the greenhouse effect, as well as the destruction of the ozone layer.

1.3.4 Other environmental effects

Ammonia in very high concentrations can be directly toxic to humans, animals and plants (Krupa, 2003). Research has shown that high atmospheric ammonia concentrations lead to an increase in N concentration in plant foliage. This in turn increases plants' sensitivity to stress, and thus makes them more sensitive to frost, drought and insect attack (Sutton *et al.*, 1993; NEGTAP, 2001). It has been suggested that foliar N content in moss could be used as an indicator of N inputs and the resulting impacts of emissions on woodland ground flora and changes in species composition (Pitcairn *et al.*, 2002). Nitrogen deposition can also affect the population of soil micro-fauna, which may slow down biological processes in the soil, especially processes that recycle nutrients such as decomposition (NEGTAP, 2001; Hornung *et al.*, 2002; Krupa, 2003).

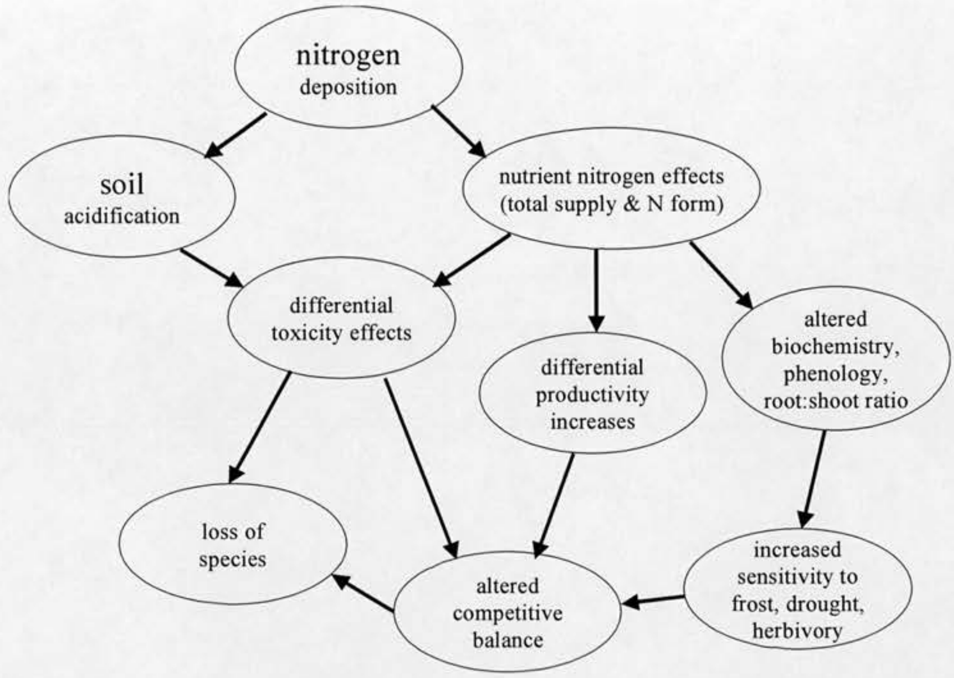

Figure 1.9. The mechanisms of effect of nitrogen deposition on vegetation. Source: NEGTAP (2001).

1.3.5 Environmental recovery

N enrichment may take place for a long time before species losses are observed, and there is often a time lag before the effects of acidification occur, due to the buffering capacity of many soils (NEGTAP, 2001; Webb *et al.*, 2002c). Although reasonably reliable estimates of the amounts of reactive N emitted to the environment are available, the rate of nitrogen being accumulated in the environment is not as well understood (Galloway and Cowling, 2002). This accumulation of N builds up large pools of reactive nitrogen in ecosystems, and today most ecosystems are assumed to be either saturated or in the accumulation phase (Krupa, 2003; Curtis *et al.*, 2005). Even sensitive ecosystems in remote areas far from any ammonia sources can be affected by reactive nitrogen through long-range transport of ammonium and oxidised nitrogen (Asman *et al.*, 1998). The National Expert Group on Transboundary Air Pollution (NEGTAP, 2001) believe it is likely that the recovery of the environment will be very slow, due to the accumulation of deposited N. They have even suggested that some of the changes may be irreversible. However, some studies have shown that recovery occurs following a decrease in atmospheric N deposition (Mitchell *et al.*, 2004).

1.3.6 Environmental effects in the UK

It is only recently that the effects of ammonia emissions in the UK have become evident. Pitcairn *et al.* (1998) showed that environmental effects can be seen in areas close to intensive farming where the ammonia dry deposition is high. When measuring ammonia concentrations close to livestock buildings, critical levels of NH_3 were exceeded for a range of plant species. Some nitrogen loving species such as *Deschampsia flexuosa, Holcus lanatus, Rubus idaeus* and *Urtica dioica* were abundant close to the livestock buildings and their percent cover decreased rapidly with distance from the source.

It has been reported that species composition has changed in some areas in the UK, with heather dominated areas becoming grass dominated, and species such as mosses, lichens and forbs have decreased relative to grasses (Pitcairn *et al.*, 1991; Thompson and Baddeley, 1991; Alonso and Hartley, 1998; Kerslake *et al.*, 1998; NEGTAP, 2001; Smart *et al.*, 2004). These changes are likely to be caused by eutrophication, acidification and/or changes in land management.

Species composition change has been reported at Rothamsted, Hertfordshire (England), where species numbers have been recorded since 1856 in a long-term experiment to study the effects of different types and amounts of fertilizers and manures on a hay meadow (Goulding *et al.*, 1998). The experiment shows a decrease in species with time (Figure 1.10), but it is difficult to conclude if this is a result of N deposition, acidification or a combination of both.

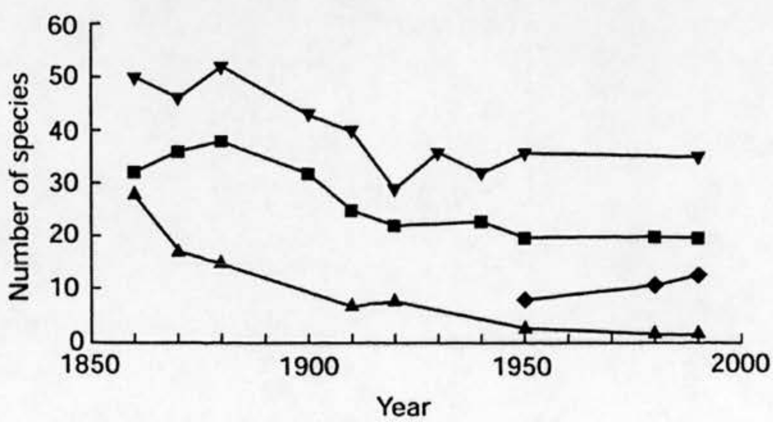

Figure 1.10. Park Grass Experiment at Rothamsted: changes in the number of species on plots. ▼ – no fertilizer, ■ – sodium nitrate, ▲ – ammonium sulphate, no lime, ♦ - ammonium sulphate with lime. Source: Goulding *et al.* (1998).-

1.4 Policy context

The increase of pollution and environmental damage worldwide has triggered the need for political action. Many pollutants are an international problem because they can be transported long distances from one country to another. This has led to the development of international agreements as a means to tackle the emission problem.

Ammonia has been identified as a major pollutant responsible for many environmental effects. In Europe there are three key international agreements dealing with ammonia emissions, the United Nations (UN-ECE) Convention on Long Range Transport of Air Pollution (CLRTAP), the EU National Emissions Ceilings Directive (NECD) and the European Directive on Integrated Pollution Prevention and Control (IPPC). The latter is important for ammonia emissions, because it regulates the pig and poultry industry, which is a major ammonia source, but the other two agreements will have a bigger effect.

Initially the political arena concentrated on one single pollutant and/or effect at a time. Sulphur was the first pollutant to be included in an emission protocol (1985), followed by protocols for NO_x (1988), VOCs (1991) and a second sulphur protocol in 1994 (NEGTAP, 2001; UNECE, 2004). In 1998 the Aarhus Protocol on Heavy Metals was adopted by CLRTAP (UNECE, 2004). More recently, a more integrated approach was adopted, because many different pollutants may interact in causing environmental effects. This resulted in *The Protocol to Abate Acidification, Eutrophication and Ground-level Ozone* (the Gothenburg Protocol) initiated by the United Nations within CLRTAP (Bull and Sutton, 1998; UNECE, 2005).

1.4.1 The Gothenburg Protocol

The Gothenburg protocol is a "multi-pollutant" as well as a "multi-effect" protocol, aimed at three effects (acidification, eutrophication and photochemical effects) and four pollutants (sulphur dioxide, oxides of nitrogen, reduced nitrogen and volatile organic compounds (VOCs)) (UNECE, 1999). The Gothenburg Protocol sets annual emission limits for the four pollutants that should be met by 2010. The target for ammonia emissions in the UK is 297 kt NH_3 per year, which is equivalent to an 11 % reduction compared with 1990 levels (NEGTAP, 2001; Webb *et al.*, 2002c). The

Gothenburg Protocol is one of eight protocols in CLRTAP and was signed in 1999 by 31 countries, including many European countries, the U.S.A. and Canada. The Gothenburg Protocol entered into force in May 2005 when the treaty had been ratified in 16 countries (UNECE, 2000).

1.4.2 NECD

The European Commissions (EC) *Directive on National Emission Ceilings* (NECD) is similar to the Gothenburg Protocol in that it targets the same pollutants and has adopted a similar approach (EC, 1999; UNECE, 1999; NEGTAP, 2001). The Gothenburg Protocol and the NECD are the first international treaties to include ammonia. The NECD requires an NH_3 emissions reduction of 21% across Europe between 1990 to 2010, while the Gothenburg protocol only requires a 12% reduction during the same time range (NEGTAP, 2001). Although the emission ceilings are stricter in the NECD than in the Gothenburg Protocol, for the UK the emission ceilings for ammonia are the same in the two emission treaties (Holland, 2001). Both the Gothenburg Protocol and NECD are expected to be reviewed in 2005-2006, assessing the expected outcome of the current policies, but also estimating the need for further action to reduce adverse effects of air pollution (Hall *et al.*, 2004b).

1.4.3 Critical levels & loads

Both the CLRTAP and the NECD aim at getting the largest possible environmental gain at the smallest possible cost. The strategy is to reduce emissions in the most cost effective way, i.e. to target countries where the emissions are most harmful and where it is relatively cheap to abate them (NEGTAP, 2001). In doing this, the two treaties have adopted *'the critical levels and loads approach'*. This approach was developed as a tool to quantify threshold levels of pollutants, below which no permanent adverse effect on the environment would be encountered. In the context of critical levels and loads, Bull (1991) refers to the term 'level' as '*the gaseous concentration of the pollutant in the air*', while the term 'load' is referred to as '*the quantity of pollutant deposited from the air to the ground*'. The critical value for the concentration (the critical level) or the deposition load (the critical load) represents the threshold where harmful effects on a receptor are expected to occur and is based on a dose-response relationship, as shown in Figure 1.11 (Bull, 1991). Receptors were defined as *'living organisms or materials which are affected, and include interrelated collections of living organisms, i.e. ecosystems'* (Bull, 1991).

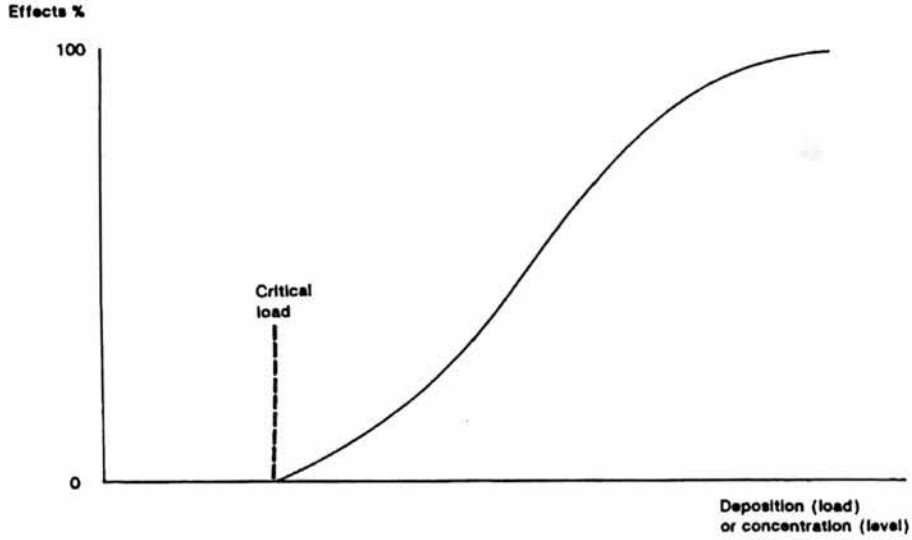

Figure 1.11. Relationship between critical load (or level) and exposure response. The threshold below which harmful effects may not occur (the critical load) is shown. Source: Bull (1991).

Critical levels

Critical levels for ammonia represent threshold atmospheric concentrations of ammonia harmful to vegetation and are dependent on the length of exposure (Table 1.2). Different receptors (species and plant communities) respond differently to NH_3 concentration, semi-natural vegetation being more sensitive than forest, with crops being least sensitive. Critical levels can be set for each individual receptor or to protect the most sensitive receptor (CLAG, 1996).

Table 1.2. Critical levels of ammonia as dependent on length of exposure. Source: CLAG (1996).

Time of exposure	Concentration above
1 day	270 $\mu g\ NH_3/m^3$
1 month	23 $\mu g\ NH_3/m^3$
1 year	8 $\mu g\ NH_3/m^3$

Critical levels for pollutants in Europe have evolved following a range of workshops. The first meeting was a UNECE workshop held in Bad Harzburg in Germany in 1988 where the following definition of critical levels was proposed (Bull, 1991):

'the concentration of pollutants in the atmosphere above which direct adverse effects on receptors, such as plants, ecosystems or materials may occur according to present knowledge.'

At the workshop, short term and long term critical levels for a range of pollutants, including ammonia, were proposed. The following year, the UNECE adopted some of these values, but the critical levels for NH_3 were not adopted, because ammonia was not considered as a transboundary air pollutant at the time (Bull, 1991). In 1992, the critical levels were revised at a Workshop in Egham in the UK (see Table 1.2), and these levels were adopted by UNECE (CLAG, 1996). These values were estimated for all vegetation types, including the most sensitive ones. In the UK, measurements have shown that critical levels of ammonia are exceeded only very close to large sources, such as immediately downwind of intensive livestock buildings (Hornung *et al.*, 2002; Pitcairn *et al.*, 2002). However, a recent assessment of nitrogen bioindicator methods, suggested that the critical level may have been set by a factor of 4 too high for some species, and hence the effect of critical levels is wider than previously estimated (Sutton *et al.*, 2004b).

Critical loads

The critical load refers to the deposited load of a pollutant and the following definition of critical loads was adopted by UNECE at the Workshop on Critical Loads for Sulphur and Nitrogen in Skokloster, Sweden, in 1988 (Nilsson and Grennfelt, 1988; Bull, 1991): *'a quantitative estimate of an exposure to one or more pollutants below which significantly harmful effects on specified sensitive elements of the environment do not occur according to present knowledge'.*

It is difficult to determine a "critical load" for ammonia, because it has both an acidification and eutrophication effect; hence for ammonia, the critical load for nitrogen and the critical load for acidity are relevant (Hornung *et al.*, 2002). If the total nitrogen deposition (derived from both oxidised and reduced N) exceeds the critical load for N, ecosystems are at risk from eutrophication, and if the critical load for acidity (derived from both N and S deposition) is exceeded, ecosystems are at risk of acidification (Hall *et al.*, 2004a). Critical loads for nutrient nitrogen were reviewed for a range of ecosystems at a UNECE workshop in Berne in November 2002 (Achermann and Bobbink, 2002). Examples of some of these critical loads are shown in Table 1.3.

Table 1.3. Examples of critical loads for nitrogen deposition (kg N ha^{-1} yr^{-1}) to mire, bog and fen habitats. ## reliable, # quite reliable and (#) expert judgement. Source: Achermann *et al.* (2002).

Ecosystem type	**Kg N ha^{-1} yr^{-1}**	**Reliability**	**Indication of exceedance**
Raised and blanket bogs	5 - $10^{a,b}$	##	Change in species composition, N saturation of *Sphagnum*
Poor fens	10 - 20	#	Increase sedges and vascular plants, negative effects on peat mosses
Rich fens	15 - 35	(#)	Increase tall graminoids, decrease diversity, decrease of characteristic mosses
Mountain rich fens	15 - 25	(#)	Increase vascular plants, decrease bryophytes

a) use towards high end of range at phosphorus limitation, and towards lower end if phosphorous is not limiting
b) use towards high end of range with high precipitation and towards low end of range with low precipitation.

EMEP, the Cooperative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe, compare critical loads maps with deposition maps to calculate areas where the critical load is exceeded in Europe. Figure 1.12 shows an example EMEP map of critical loads, indicating the sensitivity of geographical areas to pollution loads.

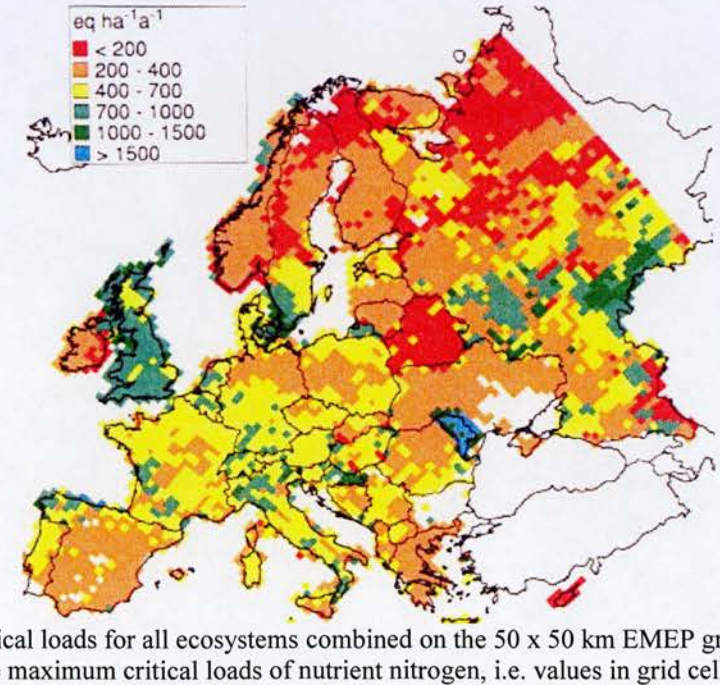

Figure 1.12. Critical loads for all ecosystems combined on the 50 x 50 km EMEP grid. The 5th percentiles of the maximum critical loads of nutrient nitrogen, i.e. values in grid cells where 95 % of the ecosystems are protected (eq $ha^{-1}a^{-1}$: equivalents per hectare per year). Source: Hettelingh *et al.* (2005).

In the UK, the National Focal Centre (NFC) for critical loads at CEH Monks Wood is responsible for calculating and mapping critical loads for UK habitats. The basic methodology to calculate critical loads maps includes two steps; first, the distribution of the main habitats is mapped, and secondly, the critical loads associated with these habitats are calculated (Hall *et al.*, 2004a). These habitats include e.g. different types of grassland and woodland, bogs (see Table 1.3), standing open water and rivers and streams. Methods to calculate critical loads can be either a 'mass-balance' approach, where the long term chemical inputs and outputs affecting acidity or nitrogen are calculated and compared with a critical chemical criterion, or an empirical approach based on expert interpretation of field studies of ecosystem response to deposition (Hall *et al.*, 2004a). Methods and data used to calculate critical loads of acidity and nutrient nitrogen in the UK are discussed in detailed in Hall *et al.* (2003) and Hall *et al.* (2004a). A critical loads exceedance calculation in the UK based on deposition data for 1995-1997 showed that 66.5 % of UK habitats are exceeded for nutrient nitrogen, and 72.6 % are exceeded for acidity (Hall *et al.*, 2004b).

Figure 1.13 shows areas in Europe where exceedance of critical loads occurs, derived by comparing critical load maps with nitrogen deposition maps. EMEP calculates deposition and critical loads exceedance maps of sulphur, nitrogen and ozone at a European scale using an Eulerian model at a grid resolution of 50 x 50 km based on the annual emission maps reported to the CLRTRAP (NEGTAP, 2001).

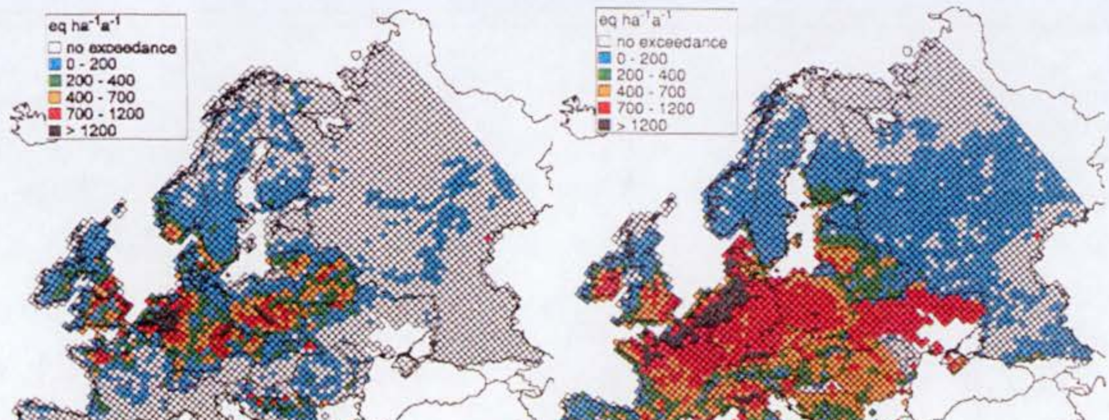

Figure 1.13. Average accumulated exceedance of a) acidity and b) nutrient nitrogen, i.e. the area-weighted average of all ecosystem exceedances in a grid cell (eq $ha^{-1}a^{-1}$: equivalents per hectare per year). Maps are calculated from critical loads of acidity and eutrophication using acid and nitrogen deposition computed by the EMEP Unified model for 2000. Source: Hettelingh *et al.* (2005).

Although the resolution in EMEP's maps has improved from the previous grid resolution of 150-km to 50-km, concern has been raised that the 50-km resolution is still too coarse to reflect variations in deposition and sensitivity within each grid square (Hirst *et al.*, 2000; Hirst and Storvik, 2003). Bertills and Lövblad (2002) highlighted that areas exceeding the critical load for eutrophication and acidification in Sweden can be as much as two to three times greater than predicted by EMEP. This suggests that adherence to the Gothenburg Protocol may be harder to achieve and that any recovery of environmentally sensitive areas will be slower than anticipated. Environmental effects, especially NH_3 effects, may occur at a much finer level of detail, and therefore there is the problem of large within-square variability of both critical loads sensitivity deposition and critical load exceedance (Sutton *et al.*, 1998; Dragosits *et al.*, 2002).

1.4.4 IPPC

In addition to the Gothenburg Protocol and NECD, the IPPC Directive (Integrated Pollution Prevention and Control) is another European Directive of importance for ammonia emission control. The directive was implemented in the UK in 2000 by the *'Pollution Prevention and Control Regulations 2000'* for industries considered to have the greatest pollution potential (SAC, 2001). In agriculture, only pig and poultry farms above a given size are affected, as they are major sources of NH_3 emissions, but the IPPC regulations for pig and poultry will not be implemented until 2006/07. The main objective of the IPPC Directive is to prevent or reduce pollution by applying best available technology (BAT). *The UK Technical Guidance* for poultry and pigs provides information on permits, and how the regulations can be fulfilled by applying BAT (IPPC, 2001, 2003).

Are the policy instruments effective?

European policy to reduce emissions has been successful in some respects. In Europe, sulphur emissions (including emissions from ships) had decreased by 56%, NO_x by 25%, NH_3 by 29% and VOCs by 40% between 1990 and 2003 (EMEP, 2005). Although some of the abated pollutants are expected to decrease substantially by 2010 and achieve the targets set in the Gothenburg Protocol, there is, however, a very different scenario for nitrogen. In terms of NH_3 contributing to the total nitrogen deposition, the prediction for the future is that emissions of ammonia are unlikely to change much between 2000 and 2010 (Figure 1.14), and that therefore areas with exceedence of critical nitrogen loads are unlikely to change much (NEGTAP, 2001).

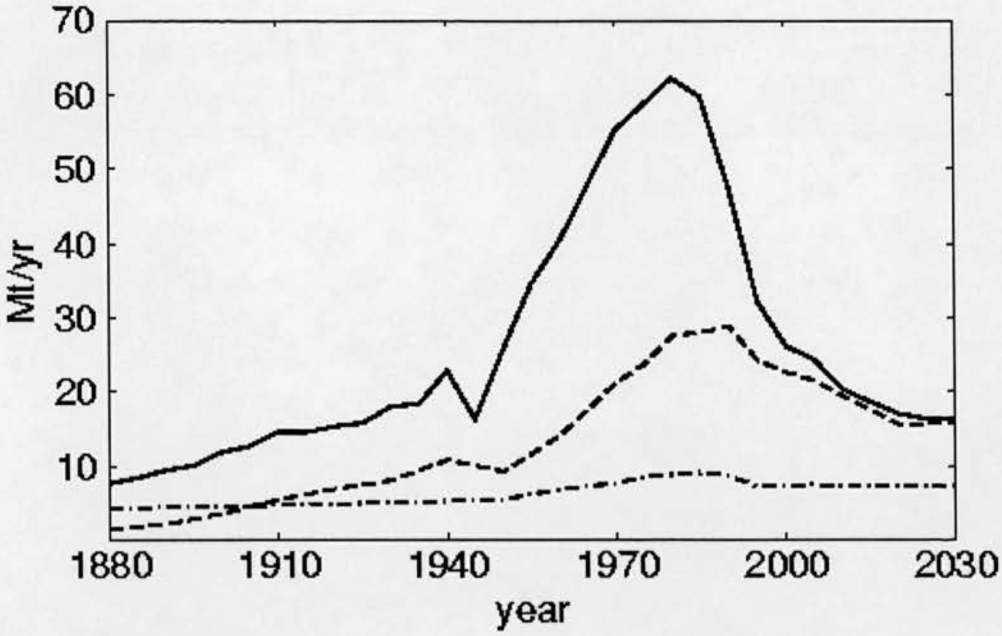

Figure 1.14 Emissions of sulfur and nitrogen in Europe over the period 1880-2030 as estimated by Schöpp *et al.* (2003). Units are Mt yr^{-1} of SO_2 (solid line), NO_2 (dashed line), and NH_3 (dot-dash line), respectively. Source: Wright *et al.* (2005).

Ammonia emissions in the UK peaked in the mid 1980s, when animal numbers were highest. Since then, emissions have decreased marginally, mainly as a result of reduced animal numbers (see Figure 1.15). Agricultural ammonia emissions in the UK have decreased by 19 % between 1990 and 2002, mainly as a result of declining animal numbers and fertilizer N use (Misselbrook *et al.*, 2003). Although NH_3 emissions have decreased in many European countries, there is still some concern that global ammonia emissions might increase in the future due to further intensification in agriculture, especially in developing countries such as China or India (Galloway and Cowling, 2002).

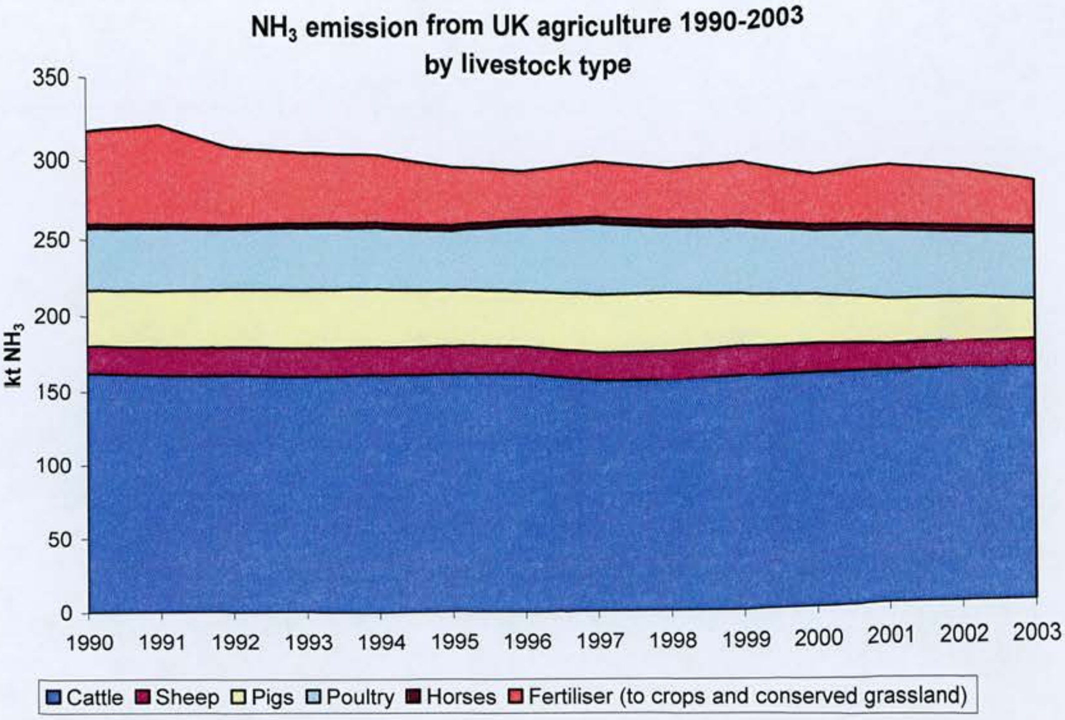

Figure 1.15. Trend in agricultural ammonia emissions in the UK 1990 – 2003, as estimated by Misselbrook *et al.* (2004).

Although the present policy instruments have a multi-pollutant and multi-effects perspective, scientists still tend to look at pollutants separately, although recently, integrated studies have started to emerge (Dalgaard *et al.*, 2002; Erisman *et al.*, 2003; Theobald *et al.*, 2004a). The problem with nitrogen is that abating one type of reactive nitrogen, e.g. NH_3, can simply take the problem elsewhere and result in e.g. increased N_2O emissions or NO_3^- (nitrate) leaching. This is due to *"the cascading effect of nitrogen"* (Galloway *et al.*, 2003), where the same atom of reactive nitrogen can cause multiple effects in different ecosystems. As a solution to dealing with this "cascading effect", some researchers have suggested the implementation of a *'Total Reactive Nitrogen Approach'* (Galloway and Cowling, 2002; Erisman *et al.*, 2003).

1.5 Emission Inventories

In an emissions inventory, the emissions of all known sources of a pollutant are estimated. In a spatial emission inventory, the magnitude of emissions is estimated and located geographically, i.e. spatially disaggregated.

1.5.1 Methodology for calculating emissions

The basic methodology to calculate emission inventories is by multiplying an emission source with its potential emission (the emission potential, also referred to as the emission factor). If the emission source is spatially distributed, a spatial emission inventory can be obtained. Emission potentials are based on research and measurements of the sources and they tend to be average values representing the typical emission for a particular source. Average emission potentials are normally very generalized, but in reality, the actual emission from a source may vary substantially due to local conditions and temporal variability.

Emission potentials can be expressed either as mass ammonia emitted per unit of activity (The Unit Approach) or as a percentage value (The Mass Flow Approach) (Webb *et al.*, 2002a). In both approaches, emission source activity data, e.g. number of livestock, surface area of storage facilities, time and amount of manure spread, are used to calculate the ammonia emission (Webb *et al.*, 2002a). The emission potential, i.e. the total ammonia emitted from an animal, e.g. a dairy cow, during a specified period of time, e.g. annually, is equal to the sum of emissions from the animal's manure during that period. All different stages in the life cycle of manure during a year are taken into account, from grazing and housing emissions of the cow, to storage and spreading emissions of the manure. For each of these stages, the amount of nitrogen that is volatilised as ammonia is estimated and summed up.

The Unit Approach estimates emissions from each source at each stage of manure management independently and then adds them together. The Mass Flow Approach, on the other hand, is built on N "flowing" through the agricultural system between management stages, with losses occurring at each stage, expressed as percentages of the previous stage (Webb and Misselbrook, 2004). This makes it possible for changes to emissions at one stage in the manure management chain to affect the following stages.

Emission potentials expressed as percentage of total N are likely to play a major role in future emission inventories and the UK is now moving from the Unit Approach, used in the past, towards the Mass Flow Approach. A system based on the Mass Flow Approach is currently being developed in the UK, the National Ammonia

Reduction Strategy Evaluation System, NARSES (Webb and Misselbrook, 2004). See Section 4.2.2 for further details on the Mass-Flow approach as integrated in the NARSES system. In this thesis, both the Unit Approach and the Mass Flow Approach are applied, compared and discussed in Chapter 4.

1.5.2 Spatial emission inventories

Spatial emission inventories have become increasingly important during the past decade, because of the need to evaluate a pollutant's effect on the environment. The rapid development in computing power over the past decade has allowed the modelling of emissions at a fine scale to become a reality. The major driving force for compiling spatial emission inventories has been scientific concern regarding the environmental effects of such emissions and the resulting political incentives to comply with international emission treaties. If the political will to develop abatement procedures is weak, the development of emission inventories is limited. This is the case outside Europe, with the U.S.A. only making a small number of limited assessments (Anderson *et al.*, 2003).

Spatial emission inventories have many advantages. They can be used to estimate the present emission in various parts of the country, but may also be used to calculate emissions from the past, as well as possible future scenarios. Emission maps are a key input to atmospheric transport models, which simulate transport and deposition of pollutants. The results of these transport models allow trans-boundary deposition to be calculated, and when these deposition maps are combined with maps of critical loads, areas exceeding the critical levels and loads of ammonia concentration and deposition can be identified. Non-spatial emission inventories are limited in their ability to evaluate the impact of emissions on sensitive areas, as well as to simulate the effectiveness of abatement measures to reduce emissions.

EMEP member countries are required to report both the total national emission, as well as the spatial distribution of emissions in their country. The Task Force on Emission Inventories and Projections (TFEIP) has developed an *'Atmospheric Emission Inventory Guidebook'* to assist member countries in the process of developing national emission inventories (TFEIP, 2004). This book is only meant to provide guidelines, as each individual country uses different modelling methodologies and emission potentials. This makes it difficult to evaluate and compare spatial emission inventories between countries, as the applied methodology is not always transparent. However, applying a common methodology and common emission potentials in Europe would not be possible for two reasons. Firstly, the quality and type of input data available to national ammonia emission models (e.g. agricultural statistics), varies between countries. Secondly, emission potentials differ between countries due to climatic differences as well as differences in agricultural practice. For instance, using the same emission potentials in a Nordic country and in Spain would not be suitable, because the climatic conditions and the agricultural practices are so different. Individual emission potentials for each country better reflect the local conditions in the region.

The level of detail in a spatial emission inventory is dependent on the pollutant studied, the purpose of the study and data availability. Local/regional inventories can include a single emission unit (e.g. a farm), or be calculated at a parish level, a county level or even a national level, whereas global inventories may cover groups of countries, a continent or the whole world. The level of detail tends to be finer the more limited the areal extent is. Generally the spatial resolution increases from global (10 x 10 or 1 x 1 degree grid cells) to European level (50 to 150 km grid resolution) to national level (grid resolutions down to 5-km, or even 1-km). In local studies the resolution may be even finer, with grid cells as small as 10 x 10 m.

1.5.3 Inventories of ammonia emission in the UK

A large number of ammonia emission inventories has been produced in the UK in the past (Table 1.4). Different methodologies have been applied, and the emission potentials used have also varied significantly. Initially, emission potentials for some sources were based on research conducted in other countries, e.g. the Netherlands, where the problem of ammonia from agriculture was highlighted at an early stage, and agriculture tends to be more intensive than in the UK. As mentioned above, applying emission potentials from other countries may not reflect the country specific conditions very well. Specific UK emission potentials have therefore been calculated that are more likely to represent UK conditions (Misselbrook *et al.*, 2000).

Table 1.4. Summary of past ammonia emission estimates in the UK. Data mainly taken from Sutton et al. (1995), with supplementary information.

Authors	Mapping resolution	Reference year	Total NH_3 (kt yr^{-1})
Healy *et al.* (1970)	National	mid 1960s	85-128[a,b]
Hood (1982)[f]	National	1978	723
Fisher (1984)	National	1977	504
ApSimon *et al.* (1987)	10-km (England & Wales)	1981	302
Buijsman *et al.* (1987)	150 km	≈ 1980	405
Ryden *et al.* (1987)	National	mid 1980s	431[d]
Thomas (1988)[g]	National	1980	488
Kruse *et al.* (1989)[h]	10-km (England & Wales) 75-km (Scotland)	1981	451
Metcalfe *et al.* (1989)[i]	100 km	1981	459-630
Jarvis and Pain (1990)	National	≈ 1988	226
Whitehead (1990)	National	Late 1980s	389-510[d]
Asman and van Jaarsveld (1992)	75 km	1987	548
Asman (1992)	75 km	1989	468
Klaassen (1992)	National	1987	492
Eager (1992)	5-km (Great Britain)	1988	409
Eggleston (1992)	National	1980-1988	527-560[b,c,j]
ECETOC (1994)	National	1990	594
Lee and Dollard (1994)	National	1991	592 (237 – 947)[a,b]
Sutton *et al.* (1995)	5-km (Great Britain)	1988	450 (231 – 715)[e]
Dragosits *et al.* (1996)	5-km (Great Britain)	1969 1988	300 287
Dragosits (1998)	5-km	1988 1996	235 233
Pain *et al.* (1998)	National	1993	197
Dragosits (1999)	5-km	1988 1996	295[e] 294[e]
Misselbrook *et al.* (2000)	National	1997	274
Misselbrook *et al.* (2003)	National	2002	251
NAEI (2004)	5-km	2002	289[e]

Note: Emission estimates included in the table are due to agricultural sources (livestock and fertilizers) except where otherwise specified. Estimates are for the whole UK unless otherwise specified. Some estimates however contain significant contributions from:

[a] Coal combustion

[b] Human sweat, sewage and other animals.

[c] Natural soils and landfill.

[d] Grassland and livestock emissions only.

[e] A comprehensive emission estimate including all known non-agricultural sources, but excluding hard standings.

[f] Quoted in Royal Society (1983).

[g] Revision of Buijsman *et al.* (1987). Estimate quoted in Bartnicki and Alcamo(1989).

[h] Estimates described in more detail by Kruse (1986).

[i] Revision of Kruse *et al.* (1989). Includes agricultural soils emissions estimated to balance atmospheric budget.

[j] Range of best estimates for different years between 1980-1988.

The use of different emission potentials in the emission inventories in the past has led to very different results, as shown in Table 1.4. The magnitude of total UK ammonia emissions in these studies cover a large range, from as little as 85 kt N yr^{-1} up to 723 kt N yr^{-1}.

The official agricultural Inventory of Ammonia Emissions in the UK (IAEUK) is calculated by Misselbrook *et al.* (2003) and updated annually. This inventory, together with an inventory of the non-agricultural NH_3 emissions (Sutton *et al.*, 2000a) is part of the UK National Atmospheric Emission Inventory (NAEI), which is the official UK ammonia emission inventory put together by organisations within the UK (CEH, IGER, AEAT etc.). The NAEI is compiled for a number of pollutants on an annual basis for submission to EMEP and other international bodies. Estimates for NH_3 from the NAEI for the years 1990 to 2003 are shown in Figure 1.16.

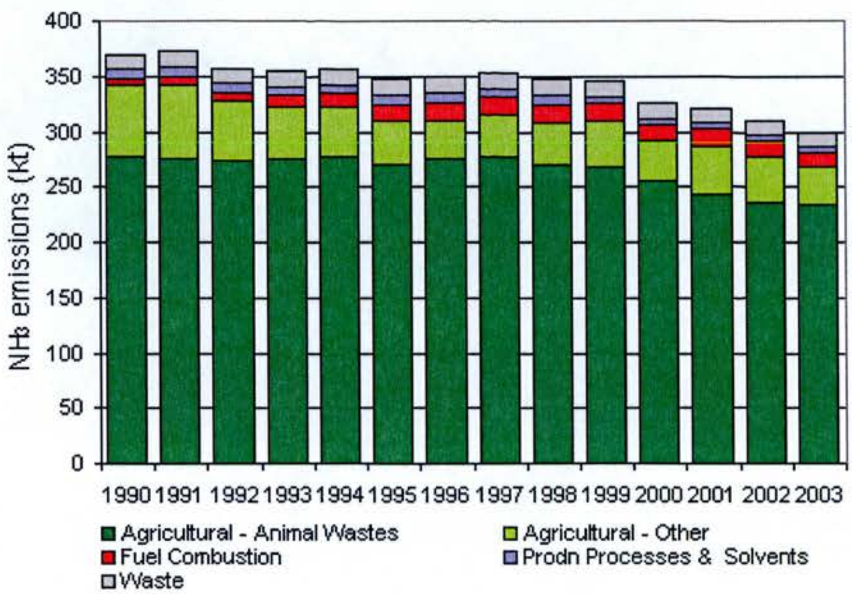

Figure 1.16. Estimated ammonia emissions in the UK from 1990 – 2003. Some non-agricultural emission sources, e.g. humans and wild animals, are excluded. Source: www.naei.org.uk.

1.5.4 Spatial distribution of ammonia emissions

Spatial ammonia emission inventories have been calculated at a global level (Schlesinger and Hartley, 1992; Dentener and Crutzen, 1994; Bouwman *et al.*, 1997), a European level (Buijsman *et al.*, 1987; Asman *et al.*, 1988), as well as at a national level in the UK and many other countries. Examples of some of the more recent national spatial ammonia emission inventories include: the UK (Dragosits, 1999), Switzerland (Rihm, 2001), Denmark (Hutchings *et al.*, 2001) and Ireland (Hyde *et al.*, 2003). Ammonia emission inventories have also been produced at a local (field) scale in many countries, e.g. in the UK (Dragosits *et al.*, 2002; Theobald *et al.*, 2004a).

The first spatially distributed ammonia emission inventory in the UK was an emission inventory for agricultural sources for England and Wales at 10 x 10 km grid resolution (Kruse, 1986; ApSimon *et al.*, 1987; Kruse *et al.*, 1989). Since then a number of spatially distributed emissions inventories have been produced for Great Britain at 5-km grid resolution (Eager, 1992; Sutton *et al.*, 1995; Dragosits *et al.*, 1996). The first spatial ammonia map for the UK was also at 5-km grid resolution (Dragosits *et al.*, 1998; Dragosits, 1999).

Modelled spatial inventories offer a relatively cheap and straightforward way to obtain ammonia emission maps for the UK. If emission values for the whole of UK were to be obtained solely from measurements, it would be a very difficult and costly task, because the spatial and temporal variation in ammonia concentration and deposition is large. Atmospheric NH_3 concentrations are very variable at a local scale due to the many rural NH_3 sources, as well as the short residence time of NH_3 in the atmosphere, as a consequence of its rapid dry deposition rate and reaction to form NH_4^+ (Sutton *et al.*, 1993; Sutton *et al.*, 1998). Measured ammonia concentrations can however be used to validate modelled ammonia concentration and deposition maps.

In the UK, the National Ammonia Monitoring Network (NAMN) was established in September 1996 and since then, consistent measurements of both NH_3 and NH_4^+ concentrations have been recorded (Figure 1.17), allowing spatial patterns of ammonia concentrations to be measured at a country scale (Sutton *et al.*, 2001b; Sutton *et al.*, 2003). Currently (December, 2005) the ammonia network consists of 95 sites (Netty van Dijk, CEH, pers. comm., 2005)

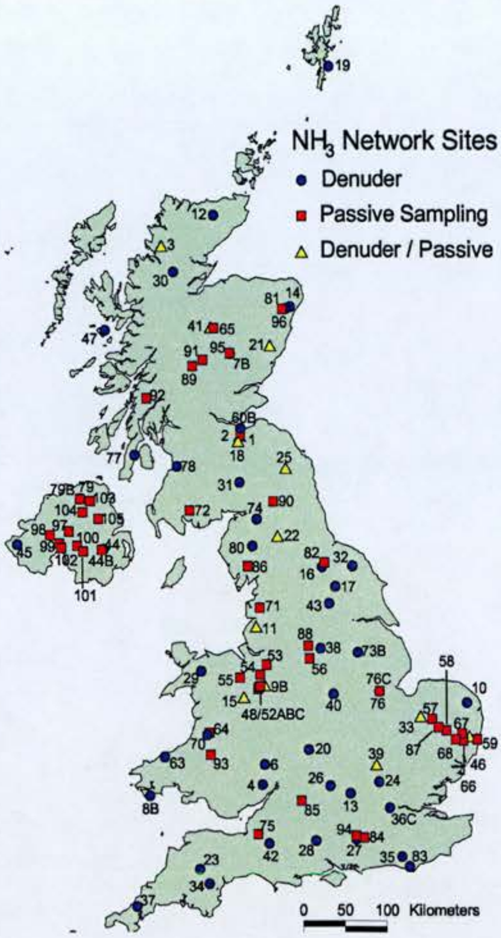

Figure 1.17. Map of sites in the UK National Ammonia Monitoring Network using either denuders, triplicate passive samplers or both (Sutton *et al.*, 2003).

Modelling methodologies and the corresponding emission potentials have evolved with time, and today thc modcl for Atmospheric Emissions for National Environmental Impacts Determination (AENEID) is a well-established tool to calculate spatial emissions of ammonia in the UK. The AENEID model was developed in 1995, and a spatially distributed UK ammonia emission inventory for year 1996 (see Figure 1.18) was subsequently compiled (Dragosits, 1999). Firstly the model distributes the emission sources across the landscape, and secondly, it assigns an emission potential to these sources. The AENEID model distributes ammonia emissions from livestock as individual sources (e.g. livestock grazing, housing, storage and spreading of manures) at the most likely location using landcover data. The main difference compared with earlier spatial inventories is in the way agricultural emission sources were redistributed in the landscape from agricultural census data.

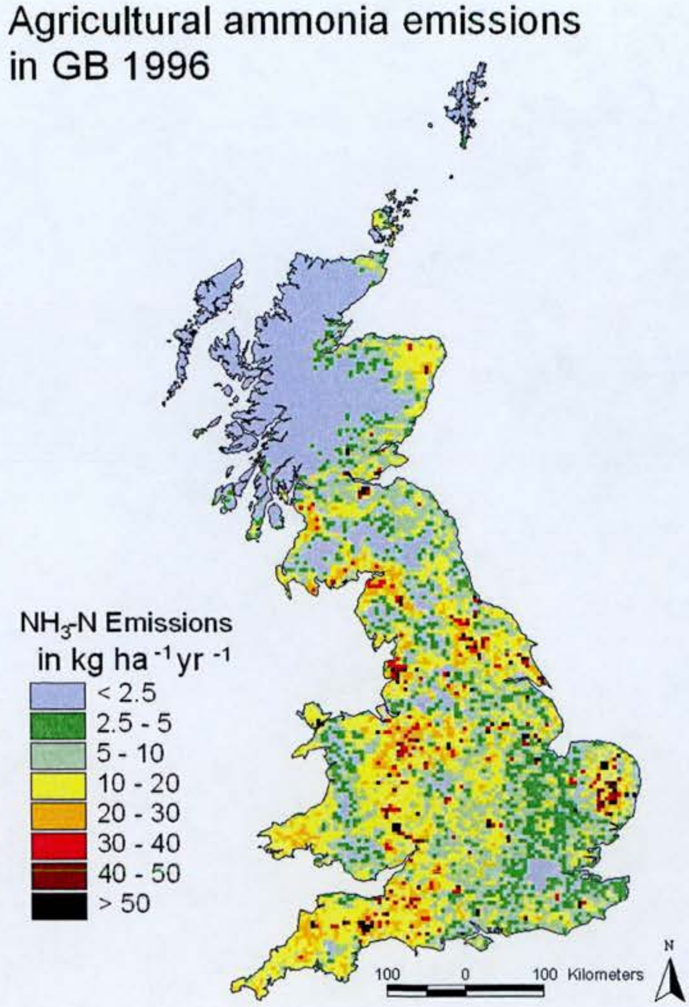

Figure 1.18. Map of total agricultural ammonia emissions for the GB (1996). Calculated using the AENEID model (Dragosits *et al.*, 1998).

Although the initial AENEID model provided a substantial improvement compared with previous spatial NH_3 emission models, Dragosits (1999) identified several areas where the model could be improved further, both regarding the spatial distribution of NH_3 sources and the spatial and temporal variability of NH_3 emission potential estimates:

1. *A strong need for a spatial process-based model of emission potential*
2. *Estimates of the intra-annual variability of NH_3 emissions*
3. *Development of a sub-model allowing manure removal from the parish of origin for large pig and poultry operations.*

One of the major uncertainties highlighted by Dragosits (1999) was the limitation of using the same average annual emission potentials for the whole of the UK. In reality emission potentials are likely to vary due to differences in farming practice and environmental factors. Dragosits (1999) therefore suggested applying regional emission potentials as a key objective for future work. Emission potentials also vary with time, so applying temporal emission potentials, e.g. on a monthly basis, would improve the modelling result even further.

Another major uncertainty pointed out by Dragosits (1999) is in the way emissions from landspreading of pig and poultry manure were distributed. In the original model all land spreading emissions occur in the parish of origin. In reality it is likely that the manure is transported over substantial distances and the emissions should therefore be located more widespread. A major limitation is therefore that the original AENEID model concentrates emissions from pigs and poultry in some areas, which results in artificial emission peaks.

Dragosits (1999) also identified the problem of the Modifiable Areal Unit Problem (MAUP). This occurs when point data are aggregated, such as the agricultural census data, are aggregated to larger spatial zones such as parishes (Openshaw and Taylor, 1981; Openshaw, 1984). Point data can be aggregated in many different ways depending on the zonal aggregation system applied, and the resulting model output is influenced by the aggregation system applied (for further details see Chapter 9). The MAUP may therefore affect the spatial location of source estimates, but the extent of the error is not currently known.

The new AENEID model presented in this thesis is a refinement of the original model developed in 1995, and many of the limitations pointed out by Dragosits (1999) have been investigated (see Chapter 3).

1.5.5 Other UK scale models of ammonia emission and deposition

The 50 x 50 km spatial resolution in the EMEP model is deemed too coarse for policies in the UK and therefore a range of other pollution models at a higher resolution are in use. The AENEID model is only one model out of a suite of models for ammonia pollution developed in the UK. FRAME, the Fine Resolution Atmospheric Multi-pollutant Exchange model, uses the AENEID ammonia emission maps as input to calculate the atmospheric concentration and deposition of NH_3 in the UK at 5-km grid resolution (Singles *et al.*, 1998; Fournier *et al.*, 2002; Fournier *et al.*, 2004). Concern has been raised that even this level of detail may be too coarse, but dispersion models at an even finer resolution have not yet been applied in the UK, due to computing limitations, uncertainties and concerns over disclosivity of input data.

NARSES, the National Ammonia Reduction Strategy Evaluation System, is a recently developed system based on the Mass Flow Approach (see Section 1.5.1 above, and Section 4.2.2) that not only estimates the total UK ammonia emissions, but also their spatial distribution as well as seasonal variation (Webb and Misselbrook, 2004). NARSES also calculates the effect and cost of potential abatement measures based on MARACCAS, the Model for the Assessment of Regional Ammonia Cost Curves for Abatement Strategies (Cowell and ApSimon, 1998). An interactive NARSES desktop model has been developed within the project, which operates at a 10 x 10 km grid resolution. In order to improve the spatial resolution of the NARSES output, the AENEID model (at 5 x 5 km resolution) has been coupled with NARSES. The AENEID model has therefore been used as part of the NARSES project (see further Chapter 4).

1.6 Thesis Plan and Research Aims

The aim of this thesis is to improve the original AENEID model to better reflect reality. This includes adding a temporal aspect, where monthly emissions are calculated as well as annual values, as well as making the model more flexible to allow the use of spatially variable emission potentials. A new modelling approach for emissions from spreading of pig and poultry manure independent of parish boundaries is developed, and uncertainties resulting from the aggregation of zones in the agricultural statistics (the Modifiable Areal Unit Problem) are assessed. The spatial emission inventory is updated to a new base year (2000) and emissions are calculated based on higher quality input data. The Isle of Man and Scilly Islands are included in the UK model for the first time.

- This chapter provided an overview of ammonia emissions, their sources, fate and impacts on the environment. The importance of spatially distributed emission inventories has been discussed, as well as the policy context of ammonia.

- Chapter 2 introduces basic processes and factors affecting ammonia emission source strength, with the main focus on agricultural sources. A brief description is also provided of how emission potentials in the UK are calculated.

- Chapter 3 describes the methodology applied in the original AENEID model to spatially distribute ammonia emissions. Areas where the model can be improved are identified and discussed.

- Chapter 4 looks at the feasibility to incorporate spatially varying emission potentials by applying emission outputs from the NARSES N-flow module to calculate emissions in the AENEID model (the coupled NARSES-AENEID approach).

- Chapter 5 gives an overview of temporal aspects affecting NH_3 emissions. Spatially distributed seasonal NH_3 emissions on a monthly basis are modelled for the year 2000 for the first time in the UK.

- Chapter 6 investigates the impact of applying a variable cattle grazing season, as the emission potential from cattle is smaller during grazing than when livestock are housed. The UK cattle grazing season is modelled for 1990, 1996 and 2000, and the spatially varying cattle emission potential is included in the AENEID model.

- Chapter 7 includes a detailed study aiming to improve the spatial distribution of manure spreading emissions from pigs and poultry. A new sub-model for pig and poultry manure spreading is developed.

- Chapter 8 discusses the results of applying the new AENEID methodology and compares the new approach with the original AENEID methodology. Inter-annual variations in ammonia emissions are also assessed for 1990, 1996 and 2000.

- Chapter 9 is a detailed study of the effect of the Modifiable Areal Unit Problem (MAUP), i.e. the impact of applying zonal aggregation of agricultural census data in an NH_3 emission inventory.
- Chapter 10 provides an overall discussion of the new AENEID model and other findings in this thesis.

1.7 Summary

Ammonia emissions to the atmosphere have increased during the last century, mainly due to the intensification of agriculture. Once ammonia has been emitted to the atmosphere, it can be transported and deposited elsewhere. Ammonia (NH_3) deposits close to its source through dry deposition, whereas its reaction product, ammonium (NH_4^+), can be transported longer distances and deposits mainly through wet deposition.

Ammonia is regarded as a pollutant responsible for many detrimental environmental effects, including eutrophication, acidification, species composition change and climate change. It has become apparent to scientists and decision makers that action has to be taken to reduce emissions and prevent further environmental damage. In 1999, ammonia was therefore included in an international emission treaty (the Gothenburg Protocol) for the first time. This Protocol is part of the United Nations Convention on Long Range Transport of Air Pollution (CLRTAP). The EU has also taken action to reduce ammonia emissions through the Directive on National Emissions Ceilings (NECD). Both the UN and the EU have adopted an approach based on critical levels and loads as a means to link atmospheric emissions with effects. EMEP member countries (including the UK) have to report their national emissions of ammonia annually in a spatial context, which has been a major driving force for the development of spatial emission inventories in Europe.

Spatial inventories play an important role when implementing abatement measures to reduce emissions. They not only make it possible to identify areas and sources of present emissions, but also to calculate emissions from the past as well as future scenarios.

Several spatial emission inventories have been calculated in the past for the UK. In this study, the aim is to provide a new, updated spatial ammonia emission inventory for the UK, building on the established AENEID model. Monthly ammonia emissions are calculated for the first time. Uncertainties in manure spreading emissions from pigs and poultry and the effect of zonal aggregation of agricultural statistics are investigated. Ways to remedy these uncertainties are taken onboard in the new AENEID model to calculate a spatial emission inventory that more closely reflects reality.

2 Processes and factors affecting ammonia emission source strength in the UK

2.1 Introduction

When ammonia emission maps are calculated, it is essential to know the emission source strength of each source as well as its spatial location. The emission source strength can also be referred to as the emission potential, and it represents the amount of ammonia emitted per source unit. NH_3 source units can be represented by both agricultural sources such as livestock and grassland or non-agricultural sources such as wild animals and industry. The main focus in research on ammonia emission potentials has been on agricultural sources, while non-agricultural sources have received less attention.

In most NH_3 emission inventories, emission potentials for individual source categories have been applied as annual national averages, but in reality, NH_3 emissions can vary significantly both in time and space. The magnitude of ammonia emissions may be affected directly or indirectly by environmental factors such as climate, topography or soil quality, or by variations in farming practice and farm type. Trying to implement all of these factors in a mathematical model is not an easy task.

In this chapter, emission potentials for ammonia are explored, with the main focus on agricultural sources. An overview of factors and processes involved in ammonia emissions is presented. In Section 2.2, basic ammonia volatilisation processes from manure are explained. Ammonia losses from different animal husbandry stages, such as grazing, livestock housing, manure storage and spreading of manure are discussed in Section 2.3. Section 2.4 explores the impact of environmental factors on ammonia emissions. Section 2.5 discusses emission potentials from agriculture and finally, emission potentials for non-agricultural sources are discussed in Section 2.6.

2.2 Basic processes of ammonia volatilisation (from manures)

In agriculture, ammonia losses occur mainly from animal manures. In the UK, about 73 million tonnes of cattle manure, 10 million tonnes of pig manure and 4.4 million tonnes of poultry manure are produced each year (Smith *et al.*, 2000, 2001a, b).

There are two main types of manure: farm yard manure (FYM) and slurry. FYM is dung and urine mixed with straw or other materials, while slurry consists of a mixture of faeces (excreta) and urine that may be diluted with water and may also contain small amounts of bedding material and waste feed (MAFF, 1987).

Ammonia losses from animal manure depend on the characteristics of the manure, which in turn depend on many different factors, e.g. breed and size of animal, age, diet, quantity of straw, addition of water and environmental conditions (MAFF, 1984; Briggs and Courtney, 1991). Table 2.1 shows an overview of manure characteristics for different types of livestock and animal wastes.

Table 2.1. Typical composition of (real weight) animal manures as summarised by Sommer and Hutchings (2001).

Manure	Animal	DM ($g\ kg^{-1}$)	N-tot ($g\ kg^{-1}$)	TAN ($g\ kg^{-1}$)	Uric acid-N ($g\ kg^{-1}$)	P ($g\ kg^{-1}$)	K ($g\ kg^{-1}$)	pH
Slurry	Cattle	74.23	3.95	1.63	n/a	0.63	3.46	7.20
Slurry	Pig	34.50	9.35	3.66	n/a	0.74	3.62	6.72
Slurry	Poultry	218.00	12.00	5.93	n/a	n/a	n/a	7.23
Solid manure	Cattle	181.50	4.85	1.33	n/a	1.45	3.85	7.80
Solid manure	Pig	222.00	10.45	4.40	n/a	3.70	5.25	7.70
Solid manure	Poultry	574.60	29.60	5.49	6.0	5.98	6.53	8.50
Deep litter	Cattle	261.00	5.20	0.90	n/a	1.40	9.70	8.60
Deep litter	Pig	412.00	11.20	2.80	n/a	n/a	n/a	8.90
Deep litter	Poultry	570.00	27.10	6.48	7.54	9.25	15.50	9.1
Liquid manure	Cattle	1.68	2.60	2.05	n/a	0.03	4.33	8.70

Dung and urine differ in composition and decomposition processes, with nutrients from dung being less mobile and less readily available than nutrients from urine. Many studies have shown the effect of a high protein content in animal feed resulting in higher nitrogen excretion rates in the urine, and altering the diet can therefore reduce ammonia emissions from livestock (Monteny and Erisman, 1998).

Ammonia (NH_3) and ammonium (NH_4^+) in manures are collectively referred to as TAN, Total Ammoniacal Nitrogen (Olesen and Sommer, 1993), and both of these reduced N species are lost from surfaces of ammoniacal solutions in water (Sommer and Hutchings, 2001). The proportion of ammonia and ammonium in a solution depends on the ammonia dissociation, which is strongly dependent on temperature and pH (Monteny and Erisman, 1998):

$$NH_3 + H_2O \leftrightarrow NH_4^+ + OH^- \qquad (2.1)$$

Volatilisation of ammonia is the mass transfer of NH_3 from the nitrogen containing substance to the air and depends on the equilibrium of NH_3 in the liquid and the gas phase at the boundary, as well as temperature and air velocity at the boundary (Olesen and Sommer, 1993; Monteny and Erisman, 1998).

$$NH_3\ (\text{aqueous}) \leftrightarrow NH_3\ (\text{gas}) \qquad (2.2)$$

The magnitude of the losses is dependent on type and characteristics of manure, local meteorological conditions (e.g. temperature and wind speed), as well as manure management practices (e.g. type of feeds and feeding routines, storage time in the barn, type of storage and spreading technique).

Two different processes are involved in the production of ammonia from animal wastes (Swensson, 2002):

1. The breakdown of organic nitrogen from excreta
2. Hydrolysis of urea, $CO(NH_2)_2$

The first process, breakdown of organic nitrogen from excreta, is a slow process and only contributes to a small proportion of ammonia losses from manure, while hydrolysis of urea (Equation 2.3), on the other hand, is a rapid process giving rise to large amounts of NH_3 (Swensson, 2002):

$$CO(NH_2)_2 + 3H_2O \rightarrow 2NH_4^+ + HCO_3^- + OH^- \rightarrow 2NH_3 + CO_2 + 2H_2O \qquad (2.3)$$

The rate of urea hydrolysis, i.e. the ammonium production rate, depends on the urea concentration and the so-called 'urease activity', which is in turn temperature-driven (Monteny and Erisman, 1998). The resulting ammonium is either oxidised by nitrifying bacteria to nitrite and nitrate, or lost through volatilisation as ammonia (Briggs and Courtney, 1991; Swensson, 2002).

Poultry excreta have a somewhat different composition than cattle and pig manures. Instead of excreting urea, poultry excrete uric acid, which is broken down to urea under moist conditions and then hydrolysed to ammonium according to Equation 2.3 (Webb *et al.*, 2002c).

The main variables controlling ammonia volatilisation from manure are summarised in Table 2.2. One of the most important factors is the TAN concentration of the manure, with greater emissions occurring from manure with high concentrations of TAN (Svensson and Ferm, 1993). The C/N-ratio of the manure may also be of importance, because ammoniacal species are bound in the strawy solid manure, and microbes will mineralise nitrogen to organic compounds if the C:N ratio is high, therefore decreasing the NH_3 losses (Phillips *et al.*, 1999). Other important factors controlling NH_3 volatilisation include pH and temperature. High temperatures and pH increase emissions, hence decreasing the temperature and adding acid to the slurry are two ways to reduce emissions (Olesen and Sommer, 1993; Monteny and Erisman, 1998). Another important factor is the air velocity at the boundary layer between the ammonia surface and the air, with high velocity encouraging ammonia volatilisation. Minimising the surface exposed to air, or reducing air velocities, e.g. the ventilation rate in livestock houses, may therefore reduce emissions of ammonia from manure (Monteny and Erisman, 1998).

Table 2.2. Important factors affecting NH_3 volatilisation and their effect.

Factor	Effect on the magnitude of ammonia volatilisation
TAN content	Ammonia losses **increase** with TAN concentration
Temperature	Ammonia losses **increase** with temperature
pH-value	Ammonia losses **increase** with pH
Air velocity	Ammonia losses **increase** with air velocity
C/N ratio	Ammonia losses **decrease** with C/N ratio

2.3 Agricultural management practices affecting ammonia emissions

Animal production systems are very inefficient regarding the transfer of N through the production cycle into the final product of meat, milk or wool (Galloway and Cowling, 2002). The excess N in this production cycle is excreted in dung and urine and either directly returned to the pasture during grazing, or collected as manure on the farm. Ammonia emissions from livestock manures occur during all stages of animal husbandry, i.e. during grazing, animal housing, manure storage and spreading of manure. The amount of NH_3 emitted at each of these stages is dependent on many different factors, e.g. type of livestock, length of time animals spend grazing, manure management system, type of manure storage and method of applying manure. The four animal husbandry stages, together with the average proportion of UK emissions from each stage are shown in Figure 2.1. Manure spreading emissions (42 %), and housing emissions (39 %) are greatest, with the remaining 19 % consisting of grazing emissions (12 %) and storage emissions (7 %) (Misselbrook *et al.*, 2003).

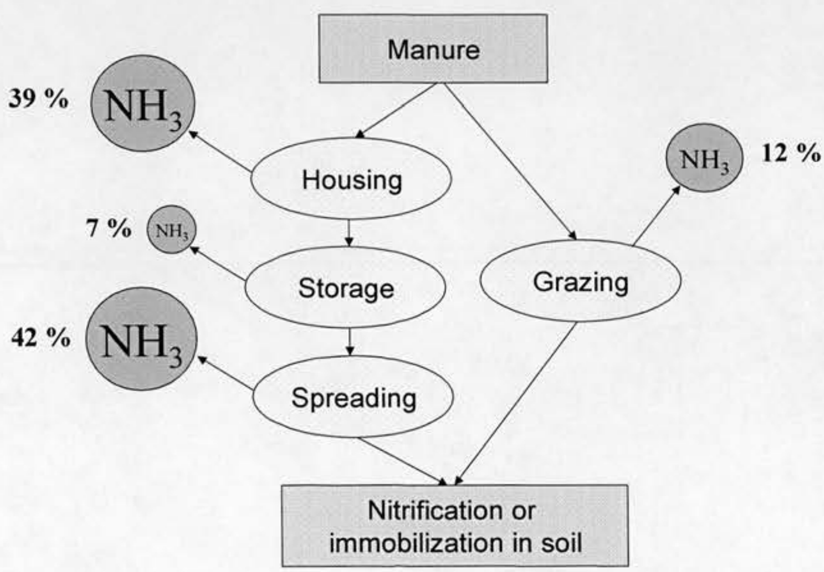

Figure 2.1. Ammonia emissions from livestock at each animal husbandry stage. Based on emission values from Misselbrook *et al.* (2003).

The proportion of emission from each of the different stages shown in Figure 2.1 represents the average value for all livestock in the UK during a year, but these percentages vary with type of livestock and time of year (Misselbrook and Smith, 2002). Cattle have greater grazing emissions and fewer housing emissions than pigs and poultry since they spend more time outdoors, mainly during the summer. These proportions may vary within each livestock sector due to differences in management practices as shown in Figure 2.2, where NH_3 emissions from hard standings, manure storage and grazing are greater from dairy cattle than from beef.

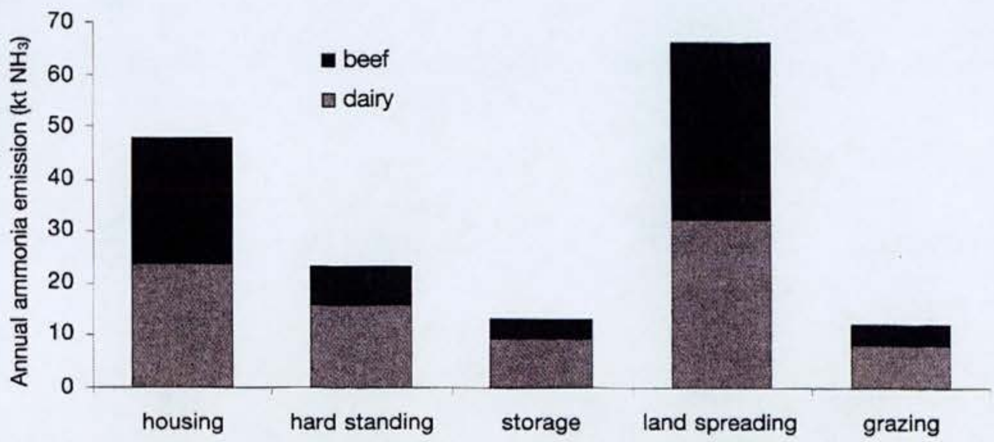

Figure 2.2. Ammonia emission estimates (kt yr^{-1}) for dairy and beef cattle farming in the UK. Source: Misselbrook and Smith (2002).

In the following sections, the main animal husbandry stages (housing, storage, landspreading, grazing and emissions from hard standings) and aspects affecting NH_3 losses during these stages are discussed.

2.3.1 Housing livestock

In a livestock building, ammonia emissions are released almost immediately when livestock urinate. As already mentioned, NH_3 volatilises from both dung and urine, and considering that adult cattle housed during winter produce 5-6 tonnes of manure per head in 6 months (MAFF, 1976), it follows that livestock buildings are associated with substantial NH_3 emissions. The amounts emitted also depend on the design and management of the housing system. Studies have shown that the ammonia concentration in livestock buildings can vary considerably within a country, even though the buildings are of the same type (Groot Koerkamp *et al.*, 1998). It is commonly accepted that aspects affecting housing emissions interact in a complex manner, and that it is difficult to predict ammonia losses from livestock buildings. The main variables affecting ammonia release from livestock buildings are summarised in Table 2.3.

Table 2.3. Main factors influencing NH_3 emissions during housing of livestock.

Factor	**Description**
Type of animal	Age, weight, size (Groot Koerkamp *et al.*, 1998).
House design and type of housing system	Animal practice, amount of litter used, labour activities and ventilation (Groot Koerkamp *et al.*, 1998).
Type of manure management system	Slurry based, straw-based, or a combination of both (Misselbrook and Smith, 2002).
Inside climate	Temperature, humidity and ventilation rate (Groot Koerkamp *et al.*, 1998).

Manure output from different types of livestock varies both in volume and manure characteristics and may even vary from livestock of the same type due to e.g. variations in age, weight and size. Groot Koerkamp *et al.* (1998) have shown that housing emissions are generally higher in poultry houses, followed by pig houses and cattle houses. In comparison, poultry emitted between 14 and 260 g NH_3 d^{-1} per (500 kg) live weight, compared with 16-90 g d^{-1} for pigs and 7.6-43 g d^{-1} for cattle.

Zhu *et al.* (2000) showed that the ventilation rate is one of the most important factors influencing ammonia emission from livestock housing. Most cattle houses in the UK are of the open "Yorkshire boarding" type, which means they are naturally ventilated (Phillips *et al.*, 2000). Estimating ammonia losses from cattle housing in the UK may therefore be difficult, because the ventilation rate in naturally ventilated houses varies with many factors such as temperature, wind speed, building design, orientation to the wind and animal occupancy.

Another important factor affecting housing emissions is the management system. Slurry-based (cubicle) systems for cattle release on average 30 % of the ammonium-N in the animal excreta, while straw-based systems only release 23 % due to absorption of some of the ammonium-N from urine in straw (Misselbrook and Smith, 2002). For pigs, the situation is reversed, and straw-based management systems release greater amounts of ammonia emissions than slurry management systems.

This is because the pig manure is less exposed to air in the slurry based system, where it falls into an underground storage tank that acts as a barrier, hence limiting contact with air. Slurry-based pig systems therefore only release about 25 % of the N content of the manure, compared with straw-based systems releasing about 40 % (Chambers *et al.*, 2002). The emission potential of poultry may also vary considerably, depending on type of housing system, as shown in Table 2.4.

Table 2.4. Ammonia emission potentials from different poultry housing systems. Source: Phillips and Chambers (2002).

Housing Type	**Average emission potential ($g\ NH_3\ LU^{-1}\ d^{-1}$)**
Laying hens (in cages with a deep pit)	170
Laying hens (in cages with belt cleaning once a week)	70
All birds kept on litter	130

Ammonia losses from livestock buildings vary diurnally, mainly as a consequence of more animal activity and higher temperatures during the day resulting in higher ammonia losses than at night (Demmers *et al.*, 1999). Housing emissions also have seasonal fluctuations, mainly because some livestock are only housed during part of the year (e.g. cattle). Even though the animals are outside grazing and not physically present in the livestock building, housing emissions may still occur because manure left in the building has a large and long-term potential for ammonia losses unless the NH_4^+ in the excreta is bound or converted. A study on empty pig farms in the Netherlands showed that the livestock buildings were still releasing ammonia, although no animals had been present in the building for 10 months (Erisman and Monteny, 1998). These pig farms were temporarily out of use because of the swine plague in the Netherlands in 1997 and manure was left in the empty pig houses. The proportion of NH_3 losses from these empty pig houses was estimated at about 20 % of the emissions from an average Dutch pig farm in operation.

2.3.2 Storing manure

Most of the manure produced by livestock in a livestock building is stored for some time before it is spread onto land for fertilization purposes. During the storage period, the composition of the manure changes due to microbial activity and organic processes (Olesen and Sommer, 1993). These processes include urea being hydrolysed to TAN, as described in Section 2.2. During storage, the concentration of TAN decreases as a consequence of volatilisation of NH_3 but also due to N_2O losses, and TAN being immobilized in bedding material (Webb, 2001). The characteristics of the manure can also change for other reasons during storage, e.g. due to precipitation, evaporation and addition of dirty water. It is also common that a crust develops on the surface of slurries during storage, which decreases NH_3 volatilisation.

Solid manure (FYM) is stackable and is often stored in open heaps with a large ammonia emitting area, compared with slurry. Ammonia losses from FYM are largest in the first few days of storage and mixing or disturbing the heap may increase the emissions further (Misselbrook and Smith, 2002). FYM is usually stored for 1 month up to 2 years, with a likely average of 6 months, while slurry storage systems are likely to emit NH_3 throughout the year as they tend to contain some waste all year around (Misselbrook *et al.*, 2000). It has however been estimated that ammonia losses from solid manure are likely to be smaller than from liquid manure during long storage periods (Dewes, 1999).

Variables influencing emissions from stored manure are summarised in Table 2.5. One of the most important factors is the surface area of the source, with smaller emissions occurring if the exposed surface area is reduced, e.g. if a crust forms on the slurry surface, or if the slurry is covered with a layer of straw (Hutchings, 1996). Slurry from cattle and pigs is normally stored in circular stores, lagoons or weeping wall stores (Misselbrook *et al.*, 2000). Above-ground circular stores are the preferred option from the perspective of ammonia emissions abatement, because it is fairly easy to cover the store and thereby reduce emissions (Nicholson *et al.*, 2002).

Table 2.5. Main factors influencing NH_3 emissions during storage of manure.

Factor	Description
Manure characteristics	TAN content and pH (Monteny and Erisman, 1998).
Type of store	Below-ground tanks, above ground circular tanks, lagoons etc. (Chambers *et al.*, 2002; Misselbrook and Smith, 2002).
Climatic conditions	Temperature and windspeed (Chambers *et al.*, 2002; Misselbrook and Smith, 2002).
Surface area of manure in contact with air	Type of storage, management practice, frequency of mixing etc. (Chambers *et al.*, 2002; Misselbrook and Smith, 2002).
Length of storage period	Total losses increase as storage time increases (Chambers *et al.*, 2002).

However, in UK farming practice not all manure is stored. About 20 % of the slurry and 50 % of FYM from cattle, and 50 % of FYM from pigs in the UK are spread directly to land without being stored (Chambers *et al.*, 2002; Misselbrook and Smith, 2002).

2.3.3 Hard standings

Hard standings are unroofed outdoor concrete yard areas, e.g. feeding yards and walkways, and emissions from these areas have generally been excluded from many ammonia emission inventories in the past. Even though hard standings may only be in use for a short time each day, emissions continue for some time once dung and urine are present. Studies have shown that the most important factors influencing this type of emission are urea concentration of the urine, urease activity, pH, temperature, air velocity and floor area, but rainfall and cleaning efficiency of the yard may also be important (Misselbrook *et al.*, 2001). It has been suggested that emissions from hard standings can be reduced by reducing the area of the yard allowed per animal, use of urease inhibitors and washing the yard instead of scraping (Misselbrook *et al.*, 2001).

Studies have shown that emissions from dairy cow feeding yards are greater than other types of yards, probably because the livestock spend more time on these yards and that the removal of dung and urine is less effective (Misselbrook *et al.*, 2001). Misselbrook *et al.* (1998) estimated that the calculated ammonia emission per dairy cow would increase by 11 % if emissions from dairy cow collecting yards were included in the IAEUK NH_3 emission inventory. It was recently agreed that emissions from hard standings would be included in the UK inventory as of the 2004 edition (T. Misselbrook, IGER, pers. comm., 2005).

2.3.4 Landspreading of manure

Manure is a valuable nutrient resource for cultivation, and is therefore spread onto agricultural land to improve crop yield and grass growth. Nitrogen in slurry and FYM is highly mobile and when animal manures have been applied to crop or grassland, the nitrogen is rapidly made available to the plants, or lost through leaching, denitrification and/or volatilisation. Most ammonia losses occur directly after spreading and thereafter gradually decline as the concentration of TAN decreases due to infiltration and volatilisation (Sommer and Hutchings, 2001), as shown in Figure 2.3. Studies have shown that NH_3 emissions following slurry application can vary by as much as 10 – 80 % of the ammonium-N applied, depending on the application method used, the characteristics of the slurry and environmental conditions (Misselbrook and Smith, 2002; Misselbrook *et al.*, 2002; Misselbrook *et al.*, 2005). N losses from slurry application normally decrease rapidly as the concentration of TAN decrease due to infiltration and volatilisation (Sommer and Hutchings, 2001). Ammonia losses from FYM spreading are lower than for slurry (approximately 65 % of the ammonium-N content of the manure), but continue for longer because the TAN does not infiltrate into the soil as easily as with slurry (Sommer and Hutchings, 2001; Misselbrook and Smith, 2002).

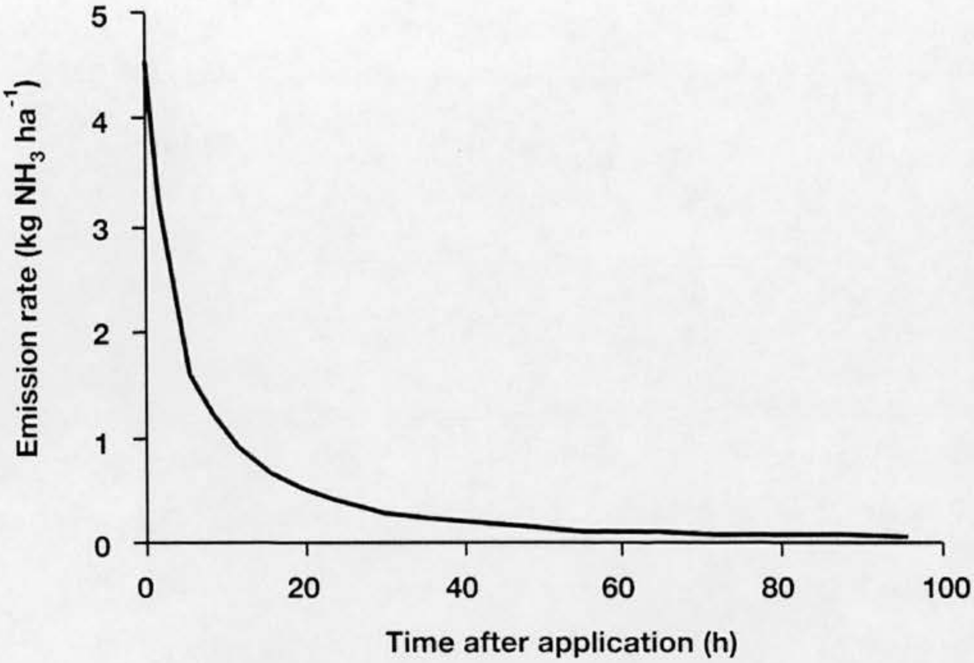

Figure 2.3. Typical temporal pattern of ammonia emissions after manure application. Source: Misselbrook and Smith (2002).

When poultry manure is spread onto land, ammonia emissions can persist for many weeks because of the slow conversion of uric acid to urea (Misselbrook *et al.*, 2000). A large amount of poultry litter in the UK is combusted annually for electricity generation, and because this litter is never spread, it is therefore not associated with any spreading emissions. Three major poultry litter power plants (Thetford, Eye and Westfield) consume 590,000 t of poultry litter per year (www.fibrowatt.com), which corresponds to about 13 % of the poultry manure production in the UK. Individual poultry farms may export as much as 95 % of their poultry litter production for incineration (M. Stevens, Grampian Country Chickens (Rearing) Ltd., pers. comm., 2003).

Factors influencing the magnitude of ammonia emissions from land spreading of manures are summarised in Table 2.6.

Table 2.6. Main factors influencing NH_3 emissions during spreading of manure.

Factor	Description
Manure characteristics	TAN content, pH and dry matter (DM) content (Sommer and Hutchings, 2001; Misselbrook *et al.*, 2005).
Storage period and losses during storage	NH_3 losses at earlier stages of manure management can influence losses at later stages (Webb and Misselbrook, 2004)
Environmental conditions during spreading	Temperature, rainfall, windspeed and solar radiation (Sommer and Hutchings, 2001; Misselbrook *et al.*, 2005).
Timing of application	Weather conditions, crop presence, sowing date and soil moisture (Sommer and Hutchings, 2001).
Method of spreading	Band-spreading, injection or incorporation (Sommer and Hutchings, 2001; Misselbrook *et al.*, 2002).

Misselbrook *et al.* (2005) identified the dry matter (DM) content as one of the most important factors influencing NH_3 emissions during spreading of particularly cattle slurry. Previous studies (Frost, 1994; Vandre *et al.*, 1997) have also shown that reducing the DM content results in smaller losses of NH_3 volatilisation, probably as an effect of a more rapid infiltration of the slurry into the soil. A reduction of the dry matter content of cattle slurry is therefore generally accepted as an effective way to reduce NH_3 losses during application of manure.

Another important factor affecting NH_3 emissions from manure spreading is the method of spreading. Where the "surface exposure" of the manure is minimized, i.e. the manure is injected or incorporated into the soil, lower emissions occur than when using traditional surface application. Hutchings *et al.* (1996) suggest that slurry injection is the most effective single method of reducing losses during spreading of slurry. Misselbrook *et al.* (2002) found that shallow injection of cattle slurry on grassland, decrease NH_3 emissions by about 73 % compared with surface broadcast. The reduction was however less efficient on arable land (23 %). Injecting or incorporating manure has the advantage that NH_4^+ is bound with the soil's cation exchange complex, which to a large extent prevents it from being volatilised (Hutchings, 1996). Although the direct ammonia emission from spreading is decreased by incorporating or injecting the manure into the soil, the additional NH_4^+-N availability in the soil may lead to N_2O emissions, NO_3^- leaching and/or stomatal emissions later on (Erisman and Monteny, 1998), see Section 2.3.6.

Environmental conditions during spreading may have a large impact on ammonia losses. Solar radiation and increased air temperature increase emissions. Rainfall events reduce the emission (particularly for slurry) due to an improved infiltration rate and the TAN being diluted. This effect is less evident for solid manure, and some studies have even shown an increase in the emission after rainfall due to the re-wetting of dried FYM, hence prolonging the length of time that NH_3 is emitted through volatilisation (Sommer and Hutchings, 2001; Misselbrook *et al.*, 2005).

The effect of temperature may be debated, as some studies have shown a relationship between NH_3 volatilisation and temperature, e.g. Sommer *et al.* (1991), while other studies have not found any significant effects of temperature during manure spreading, e.g. Braschkat *et al*, (1997) and Misselbrook *et al.* (2005). Following on from the basic processes of ammonia volatilisation discussed in Section 2.2, the theory suggests that there should be a relationship, because the processes involved in NH_3 losses are temperature dependent. This relationship has however not been evident in all experimental studies of manure application, probably due to confounding factors such as altered infiltration rate.

The effect of wind is another important environmental factor. Recently, Misselbrook *et al.* (2005) showed that wind speed was one of the most important aspect influencing NH_3 loss during spreading of cattle and pig slurry, in agreement with previous studies, e.g. Thompson *et al.* (1990).

Sommer and Hutchings (2001) summarise several different methods to decrease ammonia losses after manure spreading:

- *Application technique* – Use of trailing hoses (i.e. applying the slurry between rows of plants), or injection of slurry reduce losses, with the effectiveness of injection increasing with depth of injection.
- *Timing of application* – Best conditions to reduce losses from slurry application are during the coolest part of the day or when rain is expected.

- *Cultivation* – A cultivated soil surface reduces NH_3 losses due to a higher infiltration rate and increased surface roughness and mixing/covering with soil particles.
- *Diluting the slurry with water* – The infiltration rate may be improved, hence decreasing the emission. However, this may increase N losses through N_2O emissions and NO_3^- leaching.
- *Slurry additives* - Additives to acidify the slurry may reduce NH_3 emissions (McCrory and Hobbs, 2001).

Ammonia emissions from spreading of manure show a strong seasonal pattern. In the UK, most manure nitrogen is applied in spring to ensure rapid growth at the beginning of the growing season (Briggs and Courtney, 1991), with the exception of autumn sown crops, where the majority of the manure is applied in the autumn (Scott *et al.*, 2002). Correct timing is essential for crops to make the best use of the fertilizer. If the timing is wrong, nutrients volatilise or leach more easily to the environment. However, when the government in the Netherlands prohibited farmers from applying slurry and manure during winter (to reduce NO_3^- leaching), overall ammonia emissions may have increased (Erisman and Monteny, 1998). The reason for the increase was because farmers did not take any account of weather conditions during spreading, due to the limited application period. The agricultural land was therefore overloaded with manure during a short period of time, resulting in high levels of ammonia emission.

2.3.5 Grazing

Emissions from grasslands occur mainly during the grazing season between April to October in the UK. Grassland may be both grazed and cut, with cut grass either harvested fresh for supply to housed animals or for storage as hay or silage. Some livestock, e.g. sheep, may graze all year, while other livestock only graze part of the year (cattle) or not at all (pigs and poultry, except for free range enterprises).

Ammonia emissions during grazing derive from urine and dung being deposited on the pasture. The great majority of the nutrients taken up by the animal during grazing is returned to the field in urine and dung and either rapidly taken up by the plants, denitrified, immobilised or lost through leaching and volatilisation (Briggs and Courtney, 1991; Ross and Jarvis, 2001). Emissions from grazing animals are smaller than when livestock are housed in livestock buildings, because the urine quickly infiltrates into the soil and no additional storage, housing or spreading emissions occur (Briggs and Courtney, 1991; Misselbrook *et al.*, 2004). Total NH_3 emissions from animals grazing for part of the year are therefore dependent on the length of the grazing season. The longer the animals stay out grazing, the lower are the total NH_3 emissions during that year. Increasing the length of the grazing season could therefore reduce overall emissions from animals which are grazing for a part of the year (Webb *et al.*, 2005). Jarvis and Ledgard (2002) showed that even a small increase in the grazing period by 10 days reduced NH_3 emissions by < 4%. More recently, Webb *et al.* (2005) estimated an emission reduction of ca. 9 % for slurry-based systems and 7 % for FYM based systems, by extending the grazing season for cattle by one month. Estimating the length of the grazing season is therefore important in order to estimate ammonia emission from cattle and this is further discussed in Chapter 6. Extending the grazing season is easier further south, where the grass growing season is longer. Disadvantages are however soil compaction and trampling of sward if animals are let out too early or kept out too late.

Ammonia losses from grazing animals depend on many different factors:

- *The grass and its management,* e.g. amount of fertilizer-N applied, type of grazing system and grazing patterns (Ross and Jarvis, 2001).
- *Environmental conditions,* e.g. wind speed, air temperature, soil moisture and rainfall (Hatch *et al.*, 1990; Ross and Jarvis, 2001).
- *Length of the grazing season* (Webb *et al.*, 2005).

Grazing systems are comprised of three main components which interact closely: the livestock, the vegetation and the soil. Within this grassland-N cycle, nutrients are transported from the soil through the vegetation to the livestock and back to the soil both from the grass and the livestock (see Table 2.7). The main inputs of nutrients to a grassland system are: fertilizers, manure, soil mineralisation, atmospheric deposition and nitrogen fixation (Briggs and Courtney, 1991; Frame, 1992). The main outputs and losses are by leaching, volatilisation and the removal of grass crop and animal products. These inputs and outputs and the amount of nutrient flow vary depending on the type of grazing system. Studies have shown that the nutrient flow and ammonia losses within a grazing system increase with increased N-fertilization rate and grazing intensity (Jarvis *et al.*, 1989; Laws *et al.*, 2000). NH_3 emissions from grazing cattle are therefore related to the N input to the grassland as shown in Figure 2.4.

Table 2.7. The main components of the grassland nitrogen cycle. Source: Frame (1992).

N inputs	**Nutrient application** of organic manures, nitrate and ammonium fertilizers
	Mineralisation from soil organic matter by soil micro-organisms
	Wet and dry deposition from the atmosphere
	Symbiotic N-fixation by legume rhizobial bacteria
	Non-symbiotic N-fixation by free-living bacteria
N outputs	**Animal products,** e.g. milk, meat, wool (from grazing the sward)
	Conserved silage or hay from cut swards
N cycled	**Urine and dung** from grazing animals
	Slurry and farmyard manure from housed animals fed conserved grass
	Unutilised herbage and root tissues by senescence and soil organisms
	Nitrification of ammonium to nitrate by nitrifying bacteria in the soil
N losses	**Volatilisation of ammonia** from urine
	Immobilisation of N in soil organic matter from applied N inputs and cycled N
	Leaching of nitrate by drainage water
	Denitrification of nitrate to nitrogen gases by denitrifying bacteria in the soil
	Run-off of slurry following application in unsuitable conditions

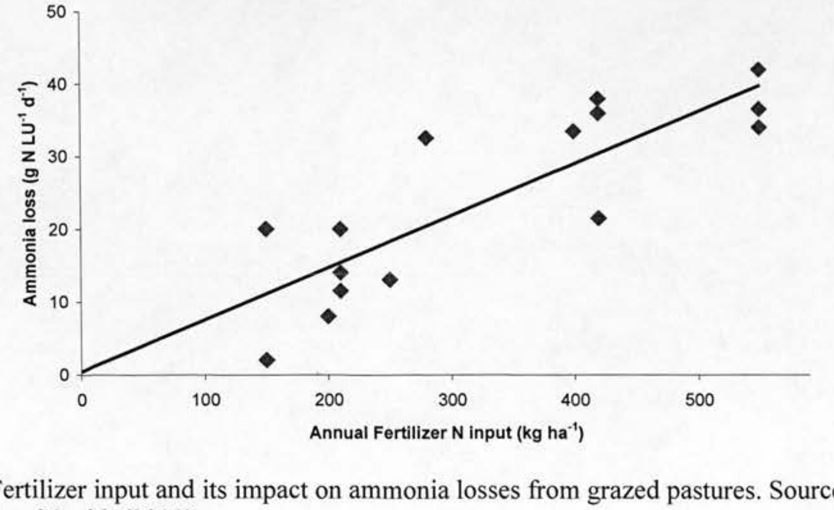

Figure 2.4. Fertilizer input and its impact on ammonia losses from grazed pastures. Source: Misselbrook and Smith (2002).

2.3.6 Fertilizers and crops

A survey in England and Wales showed that many farmers lack confidence in manures as a source of N for their crops, because manures are related to large losses of N to the environment and farmers therefore tend to trust more in inorganic fertilizers to supply sufficient nutrients to their crops (Smith *et al.*, 2001b). Ammonia losses from inorganic fertilizers are generally smaller than from animal manure (Yamulki *et al.*, 1996; Harrison and Webb, 2001). About 15 % of ammonia emissions from agriculture in the UK derive from nitrogen fertilizers (Misselbrook *et al.*, 2003), applied to cropland or grassland in order to increase yields.

The use of mineral fertilizers increased dramatically during the 20^{th} century, and according to Briggs and Courtney (1991) the use of fertilizers in Great Britain increased sevenfold between 1939 and 1975. Today the trend in fertilizer and total nitrogen use is slowly decreasing as shown in Figure 2.5. Statistics on the use of inorganic fertilizers in Great Britain for the year 2000 are shown in Table 2.8. These statistics are collected by the British Survey of Fertiliser Practice (BSFP), which collect survey data for fertilizer practice for England, Scotland and Wales on an annual basis (BSFP, 2001).

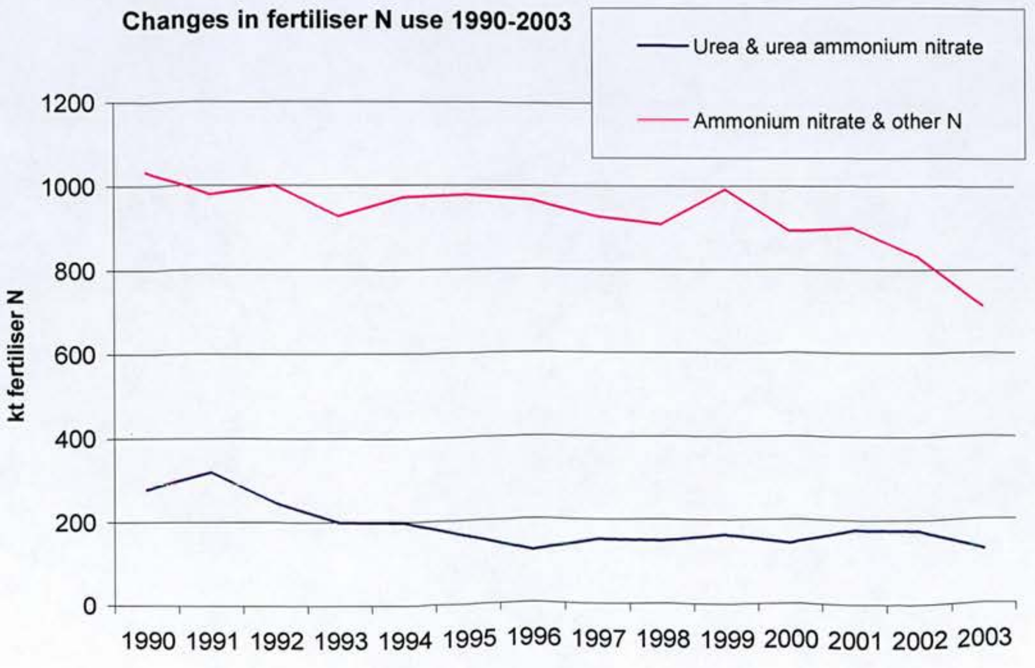

Figure 2.5. Changes in fertilizer N use 1990 – 2003. Source: Misselbrook *et al.* (2004).

column %	spring cereal	winter cereal	potatoes	sugar beet	oilseed rape	other tillage	all tillage	grass for grazing	grass for hay	grass for silage	grass not spec	all grass	all crops and grass
Calcium Ammonium Nitrate	.	0.0	.	.	.	0.2	0.0	0.0	.	.	.	0.0	0.7
Urea	0.9	5.7	0.0	1.9	11.6	2.0	5.1	1.2	2.8	1.0	1.9	1.5	3.5
Ammonium Nitrate	32.9	58.6	8.4	36.3	52.9	27.8	50.0	29.8	30.6	26.1	29.1	29.4	40.3
Other Straight N	1.5	1.4	0.5	0.3	2.6	1.0	1.4	0.6	9.5	2.8	.	1.7	1.5
Triple Superphosphate	0.2	1.1	0.3	3.2	1.4	2.5	1.2	0.2	0.3	0.2	0.9	0.3	0.8
Single Superphosphate	.	0.1	.	.	.	0.2	0.1	0.0	.	.	.	0.0	0.0
Other Straight P	0.2	0.4	0.5	1.0	0.3	3.2	0.6	0.6	0.8	0.5	.	0.5	0.6
Muriate of Potash	0.6	1.4	5.3	3.5	0.8	4.1	1.7	0.3	0.5	0.4	0.8	0.4	1.2
Other Straight K	0.1	0.1	4.3	9.2	0.2	0.4	0.7	0.3	0.4	0.2	1.0	0.3	0.5
NP	1.1	0.9	1.6	0.8	0.9	3.0	1.1	3.8	2.4	2.9	3.5	3.6	2.2
NK	1.7	0.9	.	0.6	1.2	1.3	1.0	7.2	1.8	11.8	1.2	7.1	3.7
PK	10.2	20.7	7.0	32.4	15.3	23.2	19.3	2.5	3.2	2.9	13.3	2.7	12.1
Very High N	3.4	1.7	0.1	0.0	1.3	2.9	1.8	27.9	19.4	24.0	22.5	26.9	12.8
High N	19.2	1.1	10.4	1.7	2.2	11.0	4.1	22.1	24.9	22.7	18.1	21.4	11.7
High P	1.4	0.6	5.4	0.2	0.5	2.5	1.0	0.2	0.8	0.4	.	0.3	0.7
High K	5.4	1.2	40.2	8.8	1.0	6.4	3.9	1.3	0.9	2.3	1.1	1.5	2.9
Low N	6.7	3.3	13.4	0.1	4.7	5.7	4.2	0.4	0.1	0.5	.	0.8	2.7
Low P	.	.	.	.	.	0.6	0.1	0.3	0.5	0.1	.	0.2	0.1
Equal NPK	14.4	0.9	2.6	.	3.1	2.0	2.5	1.4	1.0	1.1	6.7	1.5	2.0
Total Product ('000 tonnes)	**240**	**1621**	**104**	**108**	**249**	**212**	**2534**	**1452**	**198**	**825**	**20**	**1974**	**4508**

Table 2.8. Product type as percentage of all product used by crop group, Great Britain 2000. Source: The British Survey of Fertilizer Practice, BSFP (2001).

Ammonia emissions from fertilizers can be both direct emissions in connection with fertilizer application, or indirect emissions, as a result of foliar stomatal emissions, due to a higher nitrogen status in the vegetation caused by the fertilizer input (Sutton and Harrison, 2002). These indirect emissions may last for several weeks, while direct emissions after application of N to the field only last for a few days.

Direct emissions from fertilizers are generally well understood and extensive research has been conducted within the field (e.g. Van der Weerden and Jarvis, 1997; Harrison and Webb, 2001). Direct ammonia emissions from mineral fertilizers are small (about 0.3 – 20 % of N applied), with the exception of urea where 6 - 47% of the N applied as urea is emitted as NH_3 (Harrison and Webb, 2001; Sutton and Harrison, 2002). Changing from urea to another mineral fertilizer could therefore reduce direct emissions from fertilizers, however alternative fertilizers are generally more expensive. In the UK, only a relatively small proportion (6 %) of N fertiliser is applied as urea (Misselbrook *et al.*, 2004). Emission volatilisation rates from fertilizers are further discussed in Section 2.5.2.

Indirect emissions from fertilizers are more difficult to estimate and depend on many different factors, such as plant metabolism, environmental conditions and crop management (Sutton and Harrison, 2002). The estimates are complicated by the fact that ammonia fluxes from crops and grassland are bi-directoral, i.e. plants may act both as a source and a sink for ammonia depending on conditions (Schjoerring *et al.*, 1998). This balance is regulated by the "compensation point", which represents the balance between the ammonia concentrations in the plant's fluids and the surrounding air (Sutton *et al.*, 1993). The plant acts as a source when atmospheric concentrations are below this compensation point, and as a sink when the atmospheric concentration exceeds the compensation point. When the soil environment is changed, e.g. due to fertilizer application, or when the atmospheric conditions (e.g. increase or decrease in temperature) change the compensation point, the exchange pattern of ammonia will change too (Yamulki *et al.*, 1996). Distinct diurnal patters (Milford *et al.*, 2001) make it difficult to estimate the indirect stomatal emissions over agricultural cropland. However, studies have shown that deposition is common during cold, wet conditions, and emissions tend to dominate during warm, dry conditions (Ross and Jarvis, 2001), as shown in Figure 2.6. In general, intensive agricultural systems emit ammonia, while semi-natural ecosystems with little or no N fertilisation act as a sink to NH_3 (Sutton *et al.*, 1993; Riedo *et al.*, 2002). Agricultural land therefore tends to receive less than average deposition, while unfertilized ecosystems generally receive more than the average amount (Sutton and Fowler, 2002).

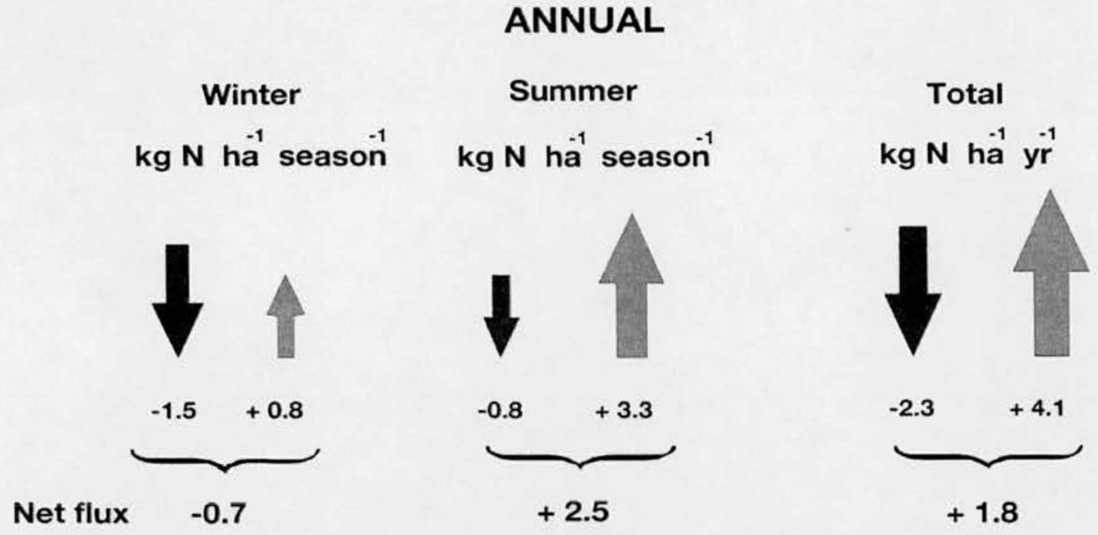

Figure 2.6. Annual fluxes of ammonia exchange in an intensely managed grassland. Source: Sutton and Harrison (2002).

Most NH_3 emission inventories only incorporate direct emissions from fertilizers and do not take indirect emissions from crop into account, because of the difficulty of estimating the emissions. As a consequence, total ammonia emissions may be underestimated. Although many studies have been conducted on the topic of stomatal ammonia emissions (Yamulki *et al.*, 1996; Asman *et al.*, 1998; Erisman and Monteny, 1998; Schjorring, 1998; Sutton *et al.*, 2000c; Milford *et al.*, 2001; Ross and Jarvis, 2001; Schjoerring and Mattsson, 2001; Huber and Kreutzer, 2002; Loubet *et al.*, 2002; Riedo *et al.*, 2002), it is difficult to estimate the net emissions. Some studies have attempted to estimate the indirect emission and come up with various estimates ranging from 1 to 4 % of the applied fertilizer N (Sutton *et al.*, 2000b; Sutton *et al.*, 2000c; Schjoerring and Mattsson, 2001).

2.4 Environmental factors affecting ammonia emissions

Environmental factors such as local climate, topography and soil quality are all important aspects that affect agricultural ammonia emissions both directly and indirectly. The UK is very diverse with large variations in environmental factors over the country, and in this section an overview of the most important environmental factors is presented.

2.4.1 Climate

Climate affects ammonia emissions in agricultural systems and significantly affects the type of agriculture practiced. Taking climatic factors into account can reduce ammonia volatilisation during land spreading of slurry, with lower emissions occurring during periods of rainfall, low temperatures and cloudy conditions (Sommer and Hutchings, 2001). Climatic factors are also responsible for the seasonality of agriculture, e.g. in determining the length of the grazing season. As already mentioned, this is important because ammonia emissions from grazing animals are lower than from housed animals and therefore affect both the magnitude of emissions as well as the temporal distribution.

The most important climatic factors in the context of NH_3 emissions are: temperature, rainfall, solar radiation, wind, relative humidity and evaporation. The relationships between all these factors and ammonia emissions are complex and not fully understood, and even if these relationships were fully understood, they would be difficult to model at a national scale due to lack of measured data.

Temperature

Temperature directly affects ammonia emissions during housing, storage, spreading of manure and fertilizer application, with higher temperatures encouraging ammonia volatilisation. Temperature is also an important factor regulating indirect NH_3 stomatal emissions from plants.

Temperature indirectly affects ammonia emissions by regulating growth, hence influencing the timing of fertilization as well as the start and length of the grazing season. Temperature therefore strongly influences the seasonal fluctuations of ammonia emissions in agriculture.

In the UK, temperature varies significantly, both with latitude and altitude, with generally lower temperatures in the upland areas. Average temperatures are therefore generally higher in the south of the UK, and decrease towards north, as shown in Figure 2.7.a.

a) **b)**

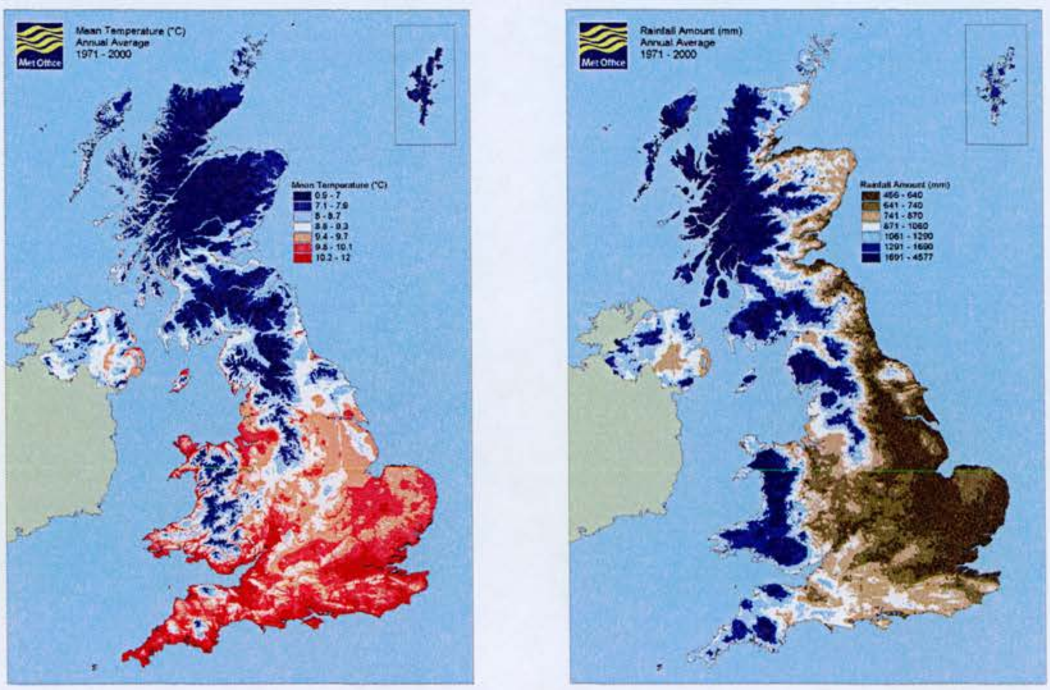

Figure 2.7. Mean annual averages in the UK 1971 – 2000 of a) temperature, and b) rainfall. Source: The Met Office, www.metoffice.gov.uk.

Precipitation

Rainfall is an important factor affecting ammonia emission, as it may increase the volume of manure during uncovered storage, by increasing the volume and diluting the TAN concentration, leading to smaller losses (Sommer and Hutchings, 2001). Rainfall may also influence NH_3 emissions during landspreading of manure and slurry. Misselbrook *et al.* (2005) showed that rainfall following application of slurry improved the infiltration of slurry TAN into the soil, resulting in lower NH_3 emissions (see Section 2.3.4 for more details).

In the UK precipitation varies far more than temperature, with average annual values ranging from only 500 mm in parts of the east to over 5000 mm in the west (Watson and Sissons, 1964). Rainfall varies considerably with altitude, and annual totals of rainfall are high in the mountainous areas and decrease gradually to the south and east, with the lowest number of rain-days found in low lying areas in central and eastern England, as shown in Figure 2.7b. The South-east of England has on average about 175 days of rain a year, while in the Highlands of Scotland it may rain for more than 250 days a year (Goudie and Brunsden, 1994). Annual values may however be misleading, because rainfall generally fluctuate seasonally. The variability of rainfall is greatest in the south-east, where as much as one third of the annual rainfall may be delivered in a single event, while it is more stable in north-western Scotland with lower variability (Goudie and Brunsden, 1994).

Effects of wind

Wind direction and velocity may affect ammonia volatilisation, as ammonia losses are encouraged by wind and increase with increased wind speed (Misselbrook *et al.*, 2005). Wind velocity affects the surface resistance of the manure, hence affecting the concentration of TAN in the surface layer (Olesen and Sommer, 1993). When NH_3 volatilises from a manure surface, the concentration of TAN in the surface layer is reduced and TAN therefore diffuses to the surface. Wind conditions during spreading of manure, as well as ventilation rates in livestock buildings, therefore have the potential to influence the magnitude of NH_3 emissions. Well-ventilated livestock buildings are likely to have greater ammonia emissions than less ventilated houses. However, the atmospheric concentrations in these buildings will be more diluted and therefore detrimental effects on the animals in these houses will be less severe (Wathes *et al.*, 2002).

Studies have shown that wind breaks around slurry stores and fields can reduce the volatilisation rate of ammonia due to the reduction in wind speed (Theobald *et al.*, 2004b). Williams (2004) estimated that NH_3 emissions from slurry stores can be reduced by about 15 %, and emissions during landspreading can be reduced by about 3 %, if wind breaks were put in place. He also showed that the benefit of windbreaks increased with decreasing source length and height of the windbreak as shown in Figure 2.8.

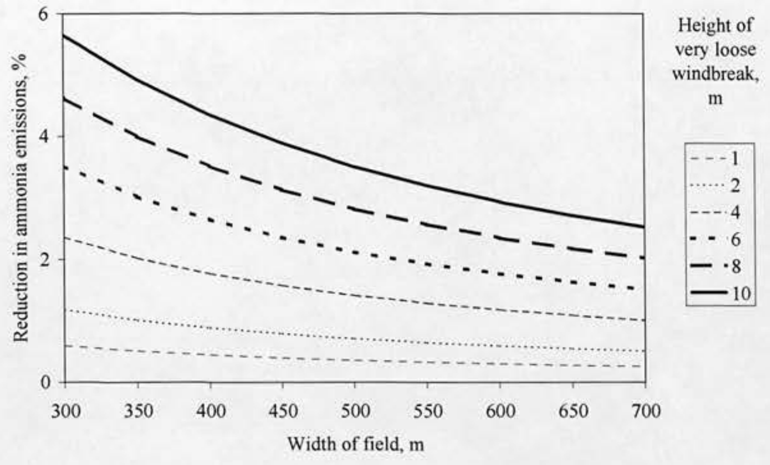

Figure 2.8. The impact of very loose windbreaks on ammonia emissions from slurry stores. Source: Williams (2004).

2.4.2 Topography

In the UK, the highest land lies in the central Highlands of Scotland and in the north west of England and Wales, and in these areas altitude and slope play an important role in agriculture. Agriculture is affected in two principal ways by relief; firstly by influencing the ease of cultivation and secondly by modifying the climate (Coppock, 1976).

Altitude strongly affects the local climatic conditions and as the altitude increases, solar radiation decreases due to increased cloud cover, mean annual temperatures decline and temperatures are more variable. Wind, relative humidity and precipitation increase at higher altitudes. These climatic changes at higher altitudes result in lower crop yields, shorter grazing seasons, increased variability of harvest and a smaller selection of crops that can be grown (Grigg, 1995). The angle and direction of slope also influence the type of agriculture, i.e. south facing slopes are more favourable for agricultural activities than north facing ones.

2.4.3 Soil quality

Soil quality in the UK can change rapidly within short distances. Soil characteristics vary considerably in structure, depth, texture, plant nutrient content and acidity and depend on factors controlling soil development, i.e. climate, parent material, vegetation cover, relief and age (Goudie and Brunsden, 1994).

Ammonia volatilisation from soils is a very complex process. Duan and Xiao (2000) studied the effects of soil properties on ammonia volatilisation and concluded that the rate of ammonia volatilisation was positively correlated with soil pH, $CaCO_3$ and total salt content, with soil pH being the most important factor. Volatilisation was negatively correlated with CEC (soil cation exchange capacity), organic matter content and clay content, with CEC being most highly correlated.

Soil type is also an important factor when determining suitable manure spreading techniques. Stoney soils make the injection of manure much more difficult, if not impossible. In such conditions bandspreading may be regarded as the best option, even though this would result in higher NH_3 emissions than when manure is injected into the soil.

2.5 Estimating emission potentials from agriculture

So far, this chapter has given an overview of the most important management practices and environmental conditions that affect the magnitude of ammonia emissions from agriculture. Studies and research into how these factors interact and affect NH_3 emissions constitute the basis for calculating emission potentials from agriculture. This review provides an indication of the complexity of including all these factors to estimate emission potentials for application in an NH_3 emission inventory.

Dragosits (1999) has presented a comprehensive overview of some of the main emission potentials applied in NH_3 emission inventories in the past. These emission potentials were derived from various institutions, including the European Centre for Ecotoxicology and Toxicology of chemicals (ECETOC), the Department of the Environment, the Task Force on Emission Inventories, the Institute of Grassland and Environmental Research (IGER), and Sutton *et al.* (1995). Applying these various emission potentials projected on livestock numbers for year 1996, Dragosits (1999) estimated an NH_3 emission range of 188.2 – 445.2 kt NH_3-N $year^{-1}$. The large range of emissions suggests that these emission potentials are associated with a great deal of uncertainty.

Today, the Institute of Grassland and Environmental Research (IGER) calculates the 'official' UK agricultural NH_3 emission inventory for Defra each year. This inventory is regarded as the most accurate agricultural ammonia inventory for the UK and is here referred to as the Inventory of Ammonia Emissions for the UK (IAEUK, Misselbrook *et al.*, 2003). The results of this emission inventory are reported in the UK National Atmospheric Emissions Inventory (NAEI), together with emission estimates for other pollutants from a variety of sources that are submitted to the United Nations Economic Commission for Europe (UNECE) under the Convention on Long-Range Transboundary Air Pollution (CLRTAP).

The IAEUK (Misselbrook *et al.*, 2004) has been calculated annually since 1996. Emission estimates have also been calculated 'backwards' until 1990, i.e. the latest emission potentials were used to recalculate emissions for all years between 1990 and the current year (Table 2.9). The inventory is updated annually both regarding activity data (statistical information) and emission potentials. The NH_3 emission potentials are updated as more research emerges and understanding of ammonia losses increases, as well as to reflect abatement strategies and other changes to agricultural management over time.

Table 2.9. Ammonia emissions (NH_3-N) from UK agriculture 1990 – 2003. Source: Misselbrook *et al.* (2004).

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
Total	262	263	251	249	249	239	239	244	237	238	223	220	212	204
Cattle	133	131	130	130	132	129	131	127	124	125	119	113	111	110
Sheep	15.3	15.2	15.4	15.5	15.4	15.3	15.0	15.1	15.6	15.7	14.9	13.1	12.8	12.8
Pigs	30.1	30.5	30.8	31.1	31.1	29.9	29.6	31.3	31.4	28.0	25.4	22.5	21.7	19.6
Poultry	33.2	33.5	32.3	33.1	32.7	31.5	34.8	38.1	34.8	35.5	33.5	35.3	33.2	33.5
Horses	2.1	2.2	2.2	2.6	2.7	2.8	3.0	3.3	3.3	2.9	3.1	3.1	3.3	3.2
Fertilizer*	48.0	51.0	40.0	36.3	34.9	30.4	25.5	29.1	27.5	31.3	27.1	32.3	30.4	25.1

**excluding fertilizer applications to grazed grassland which is included with grazing livestock*

The emission potentials applied in the IAEUK were primarily derived from studies and measurements in the UK, or when measurements were not available, from best estimates from primarily UK literature, to reflect the particular agricultural management and environmental factors typical for the UK. Examples of emission potentials for livestock and different manure management stages are shown in Table 2.10. Detailed information on how these emission potentials were derived can be found in (Misselbrook *et al.*, 2000, 2004; Webb *et al.*, 2002a).

Table 2.10. Example of UK emission potentials. Source: Misselbrook *et al.* (2004).

Livestock class	Grazing/Outdoor emission factor ($g\ N\ lu^{-1}\ d^{-1}$)	Housing emission factor [a] ($g\ N\ lu^{-1}\ d^{-1}$)		Waste storage emission factor ($g\ N\ m^{-2}\ d^{-1}$)		Land spreading emission factor[b] (% of TAN applied)	
Cattle							
Dairy cows and heifers	9.4	Dairy-cubicles	38.2	Slurry:		Slurry (based on %DM):	
Heifers in calf		-litter	25.6	Slurry stores and lagoons with crusts	4.3	Aug-Apr < 4 %	15 %
						4-8 %	37 %
				Crusted slurry stores &		> 8 %	59 %
Beef cattle	3.1	Beef-cubicles	38.2	lagoons, weeping wall stores	2.2	May-Jul all DMs	60 %
Calves	3.1	-litter	25.6	Solid manure	94 $g\ m^{-2}$ over 30 d	Solid manure	81 %
		Calves-litter	10.6	Dirty water	0.4	Dirty water	15 %
				Yard emission (dairy)	7.3 $g\ N\ animal^{-1}\ d^{-1}$		
Pigs							
Dry sows	46	Slatted	17.0	Slurry:		Slurry (based on %DM):	
		Straw bedded	16.8	Circular stores, lagoons	3.2	< 4 %	15 %
Farrowers	46	Slatted	26.6			4-8 %	37 %
		Straw bedded	26.4			> 8 %	59 %
Boars	46	Straw bedded	16.8	Solid manure	317 $g\ m^{-2}$ over 30 d		
Fatteners < 20 kg lw	101	Slatted	27.8			Solid manure	81 %
		Straw bedded	25.0				
>20-110 kg lw	53	Slatted	71.8				
		Straw bedded	68.2				
> 110 kg lw	39	Slatted	71.8				
		Straw bedded	68.2				
Poultry							
Layers	0.6 $g\ N\ birds^{-1}d^{-1}$	Deep-pit (cages, perchery, free-range)	148.8	All as solid manure heaps:	761 $g\ m^{-2}$ over 30 d	All poultry[c]	63 %
		Belt cleaned cages	60	Litter/manure	282 $g\ m^{-2}$ over 30 d		
Broilers	0.3 $g\ N\ birds^{-1}d^{-1}$	Litter	54.2				
Pullets	0.2 $g\ N\ birds^{-1}d^{-1}$	Manure	54.2				
		Litter	54.2				
Other Hens	0.6 $g\ N\ birds^{-1}d^{-1}$	Manure	54.2				
		Litter	54.2				
Other poultry	0.7 $g\ N\ birds^{-1}d^{-1}$	Litter	54.2				
Sheep							
Lowland-sheep	2.0 $g\ N\ animal^{-1}d^{-1}$	Sheep barn	4.9 $g\ N\ animal^{-1}d^{-1}$	Solid manure:	94 $g\ m^{-2}$ over 30 d	Solid manure:	81 %
-lambs	1.0 $g\ N\ animal^{-1}d^{-1}$			Hard standings:	2.8		
Upland-sheep	0.6 $g\ N\ animal^{-1}d^{-1}$						
-lambs	0.3 $g\ N\ animal^{-1}d^{-1}$						
Deer							
Deer	2.0 $g\ N\ animal^{-1}d^{-1}$	Deer barn	4.9 $g\ N\ animal^{-1}d^{-1}$	Solid manure:	94 $g\ m^{-2}$ over 30 d	Solid manure:	81 %

[a] lu is equivalent to 500 kg liveweight
[b] TAN content = total ammoniacal nitrogen content
[d] For poultry waste, TAN also includes uric acid nitrogen

2.5.1 Estimating emission potentials from livestock

There is not only a large difference in emission potential between major livestock classes, such as poultry, cattle, sheep and pigs, but differences also occur between types and breeds within each class. When Hutchings *et al.* (1996) modelled ammonia losses from a beef cattle system and a dairy cattle system, the simulated result showed over 6.5 times more NH_3 losses per dairy animal than per beef animal. These differences were mainly due to variations in management systems, feed quality, grazing patterns etc. Variations can also be related to pregnancy and lactation or age and liveweight, because younger and smaller animals have a smaller emission potential than older animals. It is therefore important to dis-aggregate national animal statistics, with each sub-class representing a livestock category with particular emission potential characteristics, so that adjustments due to variations in age, breed, management system and body size can be made.

In the present study, the emission potentials of the IAEUK 2003 (Misselbrook *et al.*, 2004) were applied for the relevant year. In the following sections, an overview of the main aspects applied to calculate these emission potentials is presented for the main livestock classes: cattle, pigs, poultry, sheep and other livestock.

Cattle

Cattle are divided into nine sub-classes based on age and type of cattle (dairy or beef cow). For each sub-class, Misselbrook *et al.* (2000) and Misselbrook *et al.* (2004) based their calculations of the emission potentials on:

- *Liveweight* - 550 kg for a milking dairy cow, 400 kg for an in-calf heifer, 140 kg for a calf and 340 kg for all other sub-classes.
- *Number of days grazing per year* – UK average figures are used, i.e. 190 days for dairy cows, 183 days for beef cattle and calves.
- *Type of housing system* - Dairy cattle are generally housed in either cubicles or on litter-based systems, beef cattle on slurry or litter-based systems and calves are all housed on litter.
- *N-inputs to grassland* - Pastures grazed by dairy cattle have a higher N input than pastures grazed by other cattle.

- *Type of manure* - Dairy cattle produce a larger proportion of slurry than other cattle, and calves are more likely to contribute to FYM rather than slurry.

Pigs

Pigs are sub-divided into five main classes: breeding sows, boars, and fatteners of different liveweight. Important factors included in the calculation of the emission potential for each sub-class are (Misselbrook *et al.*, 2000, 2004):

- *Number of pigs kept outdoors* – A larger proportion of sows (29 %) and boars (28 %) spend time outdoors compared with fatteners, of which only about 1 – 8 % spend time outdoors.
- *Type of housing system* - Boars tend to be kept on straw based systems, while sows, fatteners and piglets are kept on both straw based and on partly or fully slatted systems.
- *Type of waste* - Boars produce mainly FYM, while sows and fatteners produce both slurry and FYM.

Poultry

Poultry are divided into five sub-classes depending on type of poultry. For each sub-class, Misselbrook *et al.* (2000) and Misselbrook *et al.* (2004) base their calculation of the emission potential on the following assumptions:

- *Type of housing system* - Laying hens are generally kept in stacked cages, while broiler chickens are kept in large, undivided houses with litter on the floor. Pullets and hens are kept on either manure or litter, while other poultry, such as geese, ducks and turkey tend to be kept on litter.
- *Percentage of free-range poultry* – All broilers are kept indoors, while a proportion of layers, pullets, other hens and other poultry are estimated to be free range.
- *Manure output per bird* - Layers and other hens (the breeding flock) have a higher manure output than e.g. broilers and pullets.

Sheep

Sheep are sub-divided into lambs and lowland and upland sheep. Goats are included in the sheep category, as their emission potential is considered to be very close to that of sheep. Important factors include (Misselbrook *et al.*, 2000):

- *Number of grazing days per year* – upland sheep are considered to graze outdoors all year around, while lowland sheep are estimated to spend 30 days indoors during lambing.
- *Type of grassland* – The emission potential for lowland sheep is higher than for upland sheep due to more intense grassland management and larger nitrogen fertilizer input.

Other livestock

Other livestock include deer and agricultural horses. Emission potentials for deer were estimated for stags, hinds and calves based on number of days spent outdoors. Misselbrook *et al.* (2000) assumes that stags are outdoors all year round, while hinds and calves are outdoors for 75 % of the year. The emission potential for horses was derived from the non-agricultural emissions (Table 2.12).

2.5.2 Estimating emission potentials for fertilizers and crops

Emission potentials applied in the IAEUK for fertilizers are shown in Table 2.11, where values are expressed as % volatilisation of fertilizer N applied. Misselbrook *et al.* (2004) derive the emission potentials for ammonium nitrate and urea from Van der Weerden and Jarvis (1997), and emission potentials for UAN (urea ammonium nitrate) from the CORINAIR Inventory Guidebook (2001). Emissions from all other fertilizers are estimated to be the same as for ammonium nitrate. As already mentioned, indirect emissions from crops (stomatal emissions) have not been included in the IAEUK so far.

Table 2.11. Emission potentials (% volatilised of fertilizer N applied) for conserved grassland and tillage. Source: Misselbrook *et al.* (2004).

Fertilizer	**Grassland**	**Tillage**
Ammonium nitrate	1.6 %	0.8 %
Urea	23.0 %	11.5 %
UAN*	8.0 %	8.0 %
Other	1.6 %	0.8 %

*Urea ammonium nitrate

2.6 Estimating emission potentials for non-agricultural sources

Research on emission potentials for non-agricultural sources has not attracted as much attention as the agricultural sources, mainly because the sources are relatively small with low research priority. The most comprehensive estimates so far have been derived by Sutton *et al.* (2000a), which are now being updated annually as part of the NAEI (Dragosits *et al.*, 2004). Non-agricultural ammonia emission sources comprise a large range of different sources summarised in Table 2.12.

Table 2.12. Estimated ammonia emission potentials, and total emission from non-agricultural sources in the UK in 1996. Source: Sutton *et al.* (2000a).

Ammonia emissions from non-agricultural sources, with estimates scaled to the UK

Source	Average emission factor (units as NH_3-N)		Population (thousands)	UK emissions (kt NH_3-N yr^{-1})
Human breath	3.0 (1.0–7.7)	g $person^{-1}$ yr^{-1}	58,600	0.17 (0.06–0.45)
Human sweat	14.0 (2.1–74.9)	g $person^{-1}$ yr^{-1}	58,600	0.82 (0.12–4.39)
Infants emissions 0–3 yr[a]	13.7 (2.8–63.2)	g $infant^{-1}$ yr^{-1}	2,202	0.03 (0.01–0.14)
Cigarette smoking	17.8 (8.9–39.1)	g $smoker^{-1}$ yr^{-1}	11,251	0.20 (0.10–0.44)
Race horses	33.7 (15–40)	kg $animal^{-1}$ yr^{-1}	68[b]	2.3 (1.0–2.7)
Other horses	10.0 (5–20)	kg $animal^{-1}$ yr^{-1}	497[b]	5.0 (2.5–9.9)
Pet dogs	0.61 (0.30–0.93)	kg $animal^{-1}$ yr^{-1}	7,200	4.4 (2.1–7.0)
Pet cats	0.11 (0.05–0.16)	kg $animal^{-1}$ yr^{-1}	7,900	0.9 (0.4–1.3)
Wild deer (large)	1.23 (0.61–2.45)	kg $animal^{-1}$ yr^{-1}	471	0.6 (0.3–1.2)
Wild deer (small)[a]	0.49 (0.25–0.98)	kg $animal^{-1}$ yr^{-1}	541	0.3 (0.1–0.5)
Other major wild animals[a]	—	—	39,969	0.9 (0.2–2.5)
Large seabirds	2.15 (0.94–3.60)	kg $bird^{-1}$ yr^{-1}	648	1.4 (0.61–2.3)
Other seabirds	0.24 (0.07–0.55)	kg $bird^{-1}$ yr^{-1}	8,449	2.1 (0.62–4.7)
Biomass burning	—	—	—	1.6 (0.2–6.6)
Ecosystems		—		0
Sewage works	—	—	—	1.2 (0.7–4.9)
Sewage spreading	—	—	—	5.4 (1.5–10.2)
Landfill	—	—	—	3.3 (1.6–6.6)
Fertilizer manufacture	—	—	—	3.3 (3.3–5.0)
Sugar beet processing[a]	0.09 (0.06–0.11)	kg t^{-1} fresh beet	10.5 Mt yr^{-1}	0.9 (0.6–1.2)
Other industrial sources[a]	—	—	—	5.6 (5.6–8.4)
Transport	—	—	—	8.9 (3.3–14.5)
Domestic coal combustion	0.82 (0.41–1.65)	kg t^{-1} coal burned	2.7 Mt yr^{-1}	2.2 (1.1–4.4)
Industrial coal combustion	0.23 (0.004–4.1)	g t^{-1} coal burned	68.7 Mt yr^{-1}	0.02 (0.00–0.28)
Waste incineration	—	—	—	0.9 (0.3–2.1)
Appliances & household products[a]	—	—	—	1.0 (0.3–4.1)
Non-agricultural fertilizers[a]	—	—	—	0.2 (0.08–0.5)
Total	—	—	—	53.8 (26.9–106.4)

[a]Included for the first time in the UK inventory.
[b]British Horse Soc. (pers. comm.).

Small amounts of ammonia emissions derive from human breath, sweat, cigarette smoking and babies' nappies. Excreta from horses, dogs, cats, wild animals and seabirds, all contribute to emissions of ammonia. Most horses in the UK are used for leisure, and they are therefore treated as an additional non-agricultural source (excluding those horses reported in the agricultural census). Biomass burning is estimated to contribute to ammonia emissions, although data on biomass burning in the UK are scarce. Emissions from sewage have been calculated for wastewater treatment works as well as the application of treated sewage sludge on agricultural land, for afforestation and land reclamation purposes. Industrial ammonia emissions mainly derive from industries manufacturing fertilizers containing NH_3 and N, but also include other minor industrial sources, such as sugar beet processing. Small emissions of ammonia derive from coal combustion and waste incineration. The transport sector also contributes to ammonia emissions, mainly through catalytic converters. Household products such as garden fertilizers, hair perming solutions, cleaning solutions, latex screeding solution and refrigerants have also been estimated to emit small amounts of ammonia. Uncertainties for non-agricultural sources are high and have been estimated and ranked by Sutton *et al.* (2000a), as shown in Table 2.13.

Table 2.13. Estimated uncertainty in non-agricultural sources. Source: Sutton *et al.* (2000a).

Source	UK emissions (kt NH_3-N yr^{-1})	Range	Research priority
Sewage works & sewage spreading	6.6 (2.2-15.1)	12.9	XXXX
Transport	8.9 (3.3-14.5)	11.2	XXXX
Wild animals & sea birds	5.2 (1.9-11.2)	9.3	XXX
Horses	7.3 (3.5-12.7)	9.2	XXX
Biomass burning & ecosystems	1.6 (0.2-6.6)	6.4	XX
Pets (cats & dogs)	5.3 (2.5-8.3)	5.8	XX
Humans	1.2 (0.3-5.4)	5.1	XX
Industrial sources (inc. agro-industry)	9.9 (9.6-14.6)	5.0	XX
Landfill	3.3 (1.6-6.6)	5.0	XX
Household products & misc. fertilizers	1.3 (0.4-4.6)	4.2	XX
Coal combustion	2.2 (1.1-4.7)	3.6	X
Waste incineration	0.9 90.3-2.1)	1.8	X
Total	53.8 (26.9-106.4)	79.5	X

2.7 Discussion

Estimating ammonia emission potentials is a complicated task, because many different factors interact in a complex manner. Even if all known relationships and interactions were well known, it would still be difficult to incorporate all of them in a model due to lack of detailed data, and a large variability of emissions depending on local agricultural management and environmental factors, especially at a local level. Information on some factors e.g. factors affecting the farmers' behaviour and the way agricultural land is cultivated, are very scarce, or are generalised for the UK as a whole.

As discussed in this chapter, agricultural emission potentials are based on average farming practice and environmental conditions in the UK. In reality, two farms with the same number and type of livestock will most likely emit a different amount of ammonia due to variations in farming practice or environmental conditions. These variations in emission potentials within the UK are difficult to include in a spatial emission model, especially if they are due to "random" factors of management practice (e.g. choice of feed, housing types, N application rates etc.). Other factors that show more distinct trends, e.g. the grazing season can potentially be modelled using other variables, with known spatial distributions, such as soil type, temperatures etc.

In a non-spatial emission inventory the final result may not be influenced by regional variations in emission potential, because average emission potentials are applied and variations are evened out (unless there are non-linear relationships involved). In a spatial inventory, on the other hand, these differences may be important because they affect the spatial distribution of emissions. These types of spatial uncertainties would be reduced if regional emission potentials were applied, and if more factors influencing ammonia losses could be included in the model. So far, regional differences in the spatial distribution of NH_3 emissions have mainly been a direct result of regional differences in livestock and crop distribution. These regional differences are more emphasised if emission potentials for sub-classes of livestock (dairy cows, beef cows and calves) are applied, rather than emission potentials for livestock types (cattle, pigs, poultry and sheep). This dis-aggregation of livestock types into sub-classes is likely to increase the variability in the spatial emission inventory, as the variability in the emission potential of livestock types will not be lost by using an average value. This is likely to happen where there is a variable distribution of the sub-classes in different parts of the country. Bearing in mind that dairy farming is associated with greater NH_3 emissions than beef farming, dairy farming being more common in some areas of the UK, e.g. Shropshire, is associated in the spatial distribution maps with greater NH_3 emissions in these specialised dairy areas. If a single emission potential for average cattle was applied instead of separate emission potentials for all cattle categories of the IAEUK, this spatial variation would have been lost and resulted in overestimates in beef areas, and underestimates in dairy areas, respectively.

Estimating emission potentials is further complicated when factors counteract each other, e.g. when slurry tanks are covered to reduce emissions during storage, which results in increased emissions further down the N-flow chain (e.g. during spreading) as a consequence of more TAN being retained in the slurry. An action or change at any specific stage within the farming system will have consequences for other emission stages further down the chain (Webb and Misselbrook, 2004). These types of interactions are difficult to account for when applying traditional "average emission potentials"; they are the main driving forces behind the development of the Mass-Flow approach, where the emission potentials are applied as a proportion of TAN. In the Mass-Flow approach, the annual N excretion for each animal class is expressed in the form of TAN, and its fate is modelled through a chain of animal husbandry stages with estimates of NH_3 emissions from each stage expressed as a proportion of the available TAN (Webb and Misselbrook, 2004), as described in Section 1.5.1 and 4.2.2.

The agricultural emission potentials applied in the UK today are based on comprehensive research, and uncertainties in these values have been reduced as far as possible according to present knowledge. Agricultural emission potentials are frequently revised and updated as new research is published. Emission potentials are based on UK research output, because applying emission potentials from another country would give less reliable results, as emissions vary from country to country and even from region to region, due to differences in e.g. climate, agricultural practice and terrain. When Groot Koerkamp *et al.* (1998) compared ammonia emissions from livestock buildings in Northern Europe, emissions from UK animal housing tended to be lower than in e.g. the Netherlands, Denmark and Germany. Applying emission potentials for housing emissions from any of these countries would therefore have overestimated housing emissions in the UK.

2.8 Summary

Volatilisation rates from manure depend on the characteristics of the manure (nitrogen content, C/N ratio, pH value) as well as environmental conditions (e.g. temperature and wind velocity). Ammonia emissions from livestock occur from all stages of animal husbandry, i.e. grazing, housing animals, storing and spreading manure. The level and rate of emissions from each of these stages depend on environmental factors and farming practice, such as type of feed, livestock building, manure management system, type of storage facility and spreading technique. Environmental factors affecting ammonia emissions include climatic factors (temperature, precipitation, solar radiation and wind effects) as well as soil quality and topography.

Emissions from fertilizers may occur through both direct emissions (as a result of applying mineral fertilizers to the field) and indirect emissions (stomatal emissions). Direct emissions mainly depend on type of fertilizer and application method used, while indirect emissions are more dependent on environmental conditions, plant metabolism and crop management.

In the UK, national average emission source estimates (emission potentials) have been derived for ammonia sources, with agricultural sources receiving more attention than non-agricultural sources. These emission estimates are based on UK research and studies and are continuously updated as research within the field progresses.

3 AENEID methodology for a national ammonia emissions inventory and areas where the model can be improved

3.1 Introduction

Pollution inventories have become increasingly important as a means to evaluate environmental impacts and effects. Initially, pollution inventories were calculated without a spatial context, only showing emission values for different source types for the whole country (Jarvis and Pain, 1990; Asman, 1992). Later, the need for spatial emission inventories started to emerge, mainly driven by the need to link emissions via atmospheric transport models to environmental consequences. Spatial emission inventories are a key input to these models. Today, spatial emission inventories are common in many countries, with political incentives and emission treaties, such as the Gothenburg Protocol, acting as important driving forces.

In the early 1990's, the first spatially distributed NH_3 emission inventory for Great Britain was calculated (Eager, 1992). In 1995, the first fine-scale resolution model to spatially distribute ammonia emissions for the whole of UK was developed (Dragosits et al., 1998, 1999). The model was used to calculate emissions for 1988 and 1996 and was later named AENEID (Atmospheric Emissions for National Environmental Impacts Determination). This model and its methodology constitute the basis for the development of the new AENEID model presented in this thesis.

In this chapter, the original AENEID methodology is described and the input data to the model are introduced. Uncertainties in the methodology and input data and how they may affect the modelling results are also discussed, as well as areas where the original model can be improved.

3.2 AENEID methodology

The AENEID model is implemented as a FORTRAN77 model linked with a GIS. This implementation environment allows for the application of spatial input data and results within the GIS, while processing the data in the linked FORTRAN model.

The AENEID model calculates emissions at a 1-km level, and the resulting maps are routinely aggregated to a coarser resolution of 5-km, to reduce some of the uncertainties within the model input data and in the methodology, as well as for reasons of confidentiality regarding the detailed agricultural census data used in the model. The calculation of the NH_3 emission map is carried out in two steps (Dragosits, 1999):

- the spatial distribution of NH_3 source activities
- the assignment of NH_3 emission potentials to the sources

The basic methodology of the AENEID model is therefore to calculate an ammonia emission map by applying emission potentials to the spatially distributed source activities (see Figure 3.1). Step 3 also includes input on the emissions, as the size and type of emission activity is essential when determining the proportion of emission distributed onto different landcover types (see Section 3.4 for further details). Ammonia source activities can be either agricultural, such as livestock housing, grazing and spreading of manure and fertilizers, or non-agricultural, such as wild animals, pets, humans and industrial sources. The distribution process is described in more detail in Section 3.5.

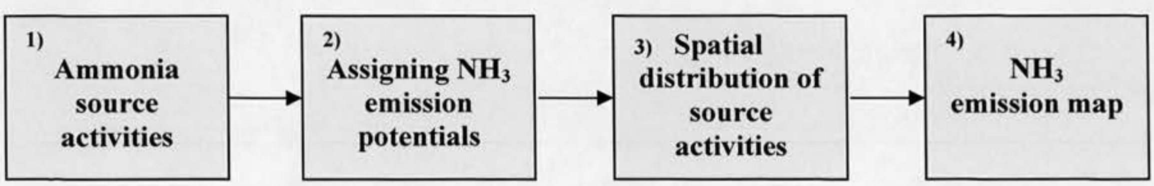

Figure 3.1. Basic methodology of the AENEID model.

3.3 Input data (agricultural emissions)

The following datasets are used as input to the AENEID model to calculate NH_3 emissions from agricultural sources:

- *Agricultural census data*
- *Landcover data*
- *The British Survey of Fertilizer Practice (BSFP)*
- *NH_3 emission potentials (source strength estimates)*

3.3.1 Agricultural Census Data

The main source of ammonia emissions is agriculture, contributing about 86 % of the total ammonia emission in the UK (NAEI, 2004). The agricultural sources can be expressed as type of livestock (calves, sows, lambs etc) and type of crops or grassland. Agricultural census statistics are therefore the most important input data to NH_3 emission inventories. The agricultural and horticultural census data are collected at the beginning of June each year and provide information about the spatial location of livestock and agricultural land in the UK. The respective government bodies within the UK are responsible for the data collection. Defra (Department for Environment, Food & Rural Affairs) collects agricultural statistics for England, the Welsh Assembly is responsible for the census data collection in Wales, the Scottish Executive for Scotland and the Department of Agriculture and Rural Development (DARD) for Northern Ireland. Although the Isle of Man is not strictly part of the UK, the ammonia emission inventory in this thesis includes the Isle of Man and these agricultural statistics were obtained from the Isle of Man Government.

In the UK, farmers are legally obliged to complete the agricultural census. The information is collected at farm holding level, but normally aggregated into larger entities for distribution due to confidentiality issues (GSS, 1997). These entities, in this thesis referred to as aggregation zones, may be of different types depending on the data provider (Table 3.1). The data can be aggregated to parish level (Scotland), grouped parishes (Wales), a 5 x 5 km grid (Northern Ireland) or the region as a whole (Isle of Man). For this study, Defra provided the English data at holding level (point data), i.e. without any aggregation, under a strict confidentiality agreement. These data were aggregated to parish level in order to make it possible to analyse the data and apply them in the model. Parishes were chosen as aggregating units in order to make the English census data aggregation uniform with the rest of Great Britain, where the Scottish data are aggregated at parish level and the Welsh data are aggregated at parish groups.

Table 3.1. Data providers and level of aggregation of agricultural census data.

	Data Provider	Aggregation level
Northern Ireland	DARD (Department of Agriculture and Rural Development)	5 x 5 km grid (1996 & 2000) Rural district level (1990)
England	Defra (Department for Environment, Food and Rural Affairs)	Holding level
Wales	Welsh Assembly Government	Parish groups
Scotland	Scottish Executive	Parishes
Isle of Man	Isle of Man Government	Isle of Man

The effect of applying different types of aggregation zones on the English holding data are investigated further in Chapter 9, where the data are aggregated at parish level, at a 1-km grid, a 5-km grid, and a 10-km grid, respectively. The study showed that the choice of aggregation zones significantly influences the emission result.

The data providers had, to some degree, collected the census data using different categories. Furthermore, the census categories were not always consistent for three different years (1990, 1996 and 2000) used in this study, even from the same data provider. In order to integrate the agricultural census data into the NH_3 emission model, the datasets had to be unified into a common set of categories. An agreed set of 46 livestock and crop categories for the different devolved regions of the UK and for the years 1990, 1996 and 2000 was therefore developed (see Table 3.2). These categories were agreed within the NARSES-framework (the National Ammonia Reduction Strategy Evaluation System), which is further explained in Section 4.3.1. Generating these 46 categories for input to the model was straightforward for most datasets as this had mostly required aggregation of existing categories. The Welsh dataset was, however, more complicated, because not all individual census categories were provided by the Welsh Assembly, but rather an aggregated version due to statistical uncertainties. The Welsh statistics do not, for example, distinguish between different types of pigs, as <u>all</u> pigs are aggregated into one category. Details on the Welsh aggregation of the agricultural statistics compared with the 46 livestock and crop categories are provided in Table 3.2. In order to integrate these aggregated categories into the AENEID model, they had to be dis-aggregated into the common 46 categories using additional statistical data. This was carried out by deriving the total value for the missing category from the relevant statistics from the Welsh Assembly and scaling it to the aggregated category provided. The total number of pigs for the different pig categories were derived from 'Welsh Agricultural Statistics 2001' (WA, 2002), e.g. total number of boars. These total numbers were then scaled to the pig-category provided by the Welsh Assembly at grouped parish level according to:

$$boars\_pg1 = boars\_stat * pigs\_pg1 \; / \; pigs\_tot \qquad (3.1)$$

where	*boars_pg1*	Number of boars in parish group 1 (calculated value)
	boars_stat	Total number of boars in Wales year 2000 (derived from 'Welsh Agricultural Statistics 2001')
	pigs_pg1	Total number of pigs in parish group 1 (provided by the Welsh Assembly)
	pigs_tot	Total number of pigs in all parish groups (provided by the Welsh Assembly)

This "reverse engineering" of the statistics compensates for errors in the magnitude of emissions due to the different emission potentials for different types of pigs (Misselbrook *et al.*, 2000). Such reverse engineering, however, does not compensate for spatial errors due to variations in the proportion of, for example, boars across the country, because the process assumes that the proportion of boars is the same across all parish groups in Wales.

The agricultural census data from the devolved regions made available for this project, were obtained under strict confidentiality agreements, and the census returns for individual holdings must not be disclosed. It should be stressed that although the *input* data for AENEID are disclosive, the *output,* i.e. the NH_3 emission maps are non-disclosive, as the production of the ammonia emission maps ensures that the contributing sources of ammonia are hidden due to the spatial distribution process.

Table 3.2. Livestock and crop categories applied in the new AENEID model, compared with the agricultural statistics provided by the Welsh Assembly. (See Section 4.3.1 for further details on the agricultural categories used as input to AENEID.)

	NARSES-Id	Description	Welsh categories provided
Cattle	1	Dairy cows & heifers	Dairy cattle
	2	Dairy heifers in calf, 2 years and over	
	3	Dairy heifers in calf, less than 2 years	
	4	Beef cows & heifers	Beef cattle
	5	Beef heifers in calf, 2yrs and over	
	6	Beef heifers in calf, less than 2 years	
	7	Bulls >2yrs	Other cattle
	8	Bulls 1-2yrs	
	9	Other cattle, over 2yrs	
	10	1Other cattle, 1-2yrs	
	11	Other cattle, under 1yr	Calves
Sheep	12	Sheep	Sheep
	13	Lambs, under 1 year old	Lambs
Pigs	14	Sows in pig & other sows	Pigs
	15	Gilts in pig & barren sows for fattening	
	16	Gilts > 50kg not yet in pig	
	17	Boars	
	18	Other pigs, 110kg and over	
	19	Other pigs, 80-110kg	
	20	Other pigs, 50-80kg	
	21	Other pigs, 20-50kg	
	22	Other pigs, under 20kg	
Poultry	23	Layers	Fowls
	24	Breeding birds	
	25	Broilers	
	26	Pullets	
	27	Turkeys	Other poultry
	28	Other poultry	
Horses	29	Horses	Horses
Goats	30	Goats	Goats
Deer	31	Deer	-
Arable Land	32	Set-aside land	-
	33	Wheat	Wheat
	34	Winter Barley	Winter Barley
	35	Spring Barley	Spring Barley
	37	Oilseed rape	Oilseed rape
	38	Potatoes	Potatoes
	39	Other cereals	Other cereals
	36	Sugar beet	Other crops
	40	Other root crops	
	41	Other crops	
	42	Vegetables for human consumption	
	43	Fruit	
	44	Bulbs, flowers and nursery stock	
	45	Grassland under 5 years	Grassland under 5 years
	46	Permanent grassland	Permanent grassland

3.3.2 Land cover data

Landcover data are essential in the AENEID model to distribute ammonia sources, such as livestock and crop, spatially within each aggregation zone, thereby linking NH_3 emissions to specific landcover and locations in the landscape. Linking source activities to a landcover dataset at 1-km resolution makes it possible to estimate emissions at this level of detail, even though the original resolution of the source activities is much coarser, i.e. at the parish level.

In the original AENEID model (Dragosits, 1999), a landcover map from 1990, the Land Cover Map of Great Britain (LCMGB) was applied. This dataset was derived from conventional classification of raster satellite images of Great Britain (LANDSAT Thematic Mapper) recorded during 1988 – 1989 (Fuller *et al.*, 1994). The landcover dataset applied in this study is the UK Land Cover Map 2000 (LCM2000). It has not only been updated, but also upgraded from LCMGB 1990 in terms of the satellite landcover classification methodology. The data were recorded a decade later (1997-2000), and a different classification approach based on a vector data structure was applied (Fuller *et al.*, 2002). LCM2000 also includes Northern Ireland, hence representing the whole of UK Furthermore, LCM2000 does not have any unclassified areas, whereas in LCMGB1990 some areas remained unclassified due to cloud cover.

The 1-km LCM2000 dataset contains percentage values for 27 different landcover classes for each gridcell. Of these 27 classes, 13 were regarded as suitable for inclusion in the AENEID model and these were aggregated into 6 relevant classes for the purpose of re-distributing agricultural sources within the aggregation zone. These six landcover classes were: arable, improved grassland, partially improved grassland, rough grassland, poor rough grazing and suburban. The aggregation key is shown in Table 3.3. The distribution of each of the 6 landcover classes in the UK is shown in Figure 3.2.a-f.

Table 3.3. Aggregation of landcover classes for the purpose of spatially distributing ammonia emissions in the UK.

Aggregated AENEID class	LCM2000 class
Arable	Arable cereals Arable horticulture Arable non-rotational
Improved grassland	Improved grassland
Partially improved grassland	Neutral grass Setaside grass Calcareous grass Acid grassland
Rough grazing	Open dwarf shrub heath Fen, marsh, swamp
Poor rough grazing	Saltmarsh Dense dwarf shrub heath
Suburban	Suburban/rural development

The Isle of Man was included in LCM2000, but not the Scilly Isles. The necessary landcover classes for Scilly Isles were therefore generated with the aid of other sources such as topographic maps.

3.3.3 Emission potentials

Once the ammonia sources (livestock categories) have been spatially distributed with the aid of landcover data, emission potentials are applied to calculate the final emission map. These emission strength estimates are expressed per source unit and are derived from the Inventory of Ammonia Emission in the UK (IAEUK) by Misselbrook *et al.* (2004). This is carried out by dividing the total emission from the source category by the total number of units in the UK, e.g. dividing the total estimated NH_3 emission for dairy cows by the number of dairy cows to arrive at the NH_3 emission estimate per dairy cow to be applied. (See Section 2.5.1. for details on how the total ammonia for each livestock category is calculated in IAEUK.) Emission potentials and their uncertainties have already been discussed in detail in Chapter 2. In this study, average NH_3 emission potentials as well as regionally varying emission potentials were applied in the AENEID model to generate spatially distributed emission maps (see Chapter 4 for details).

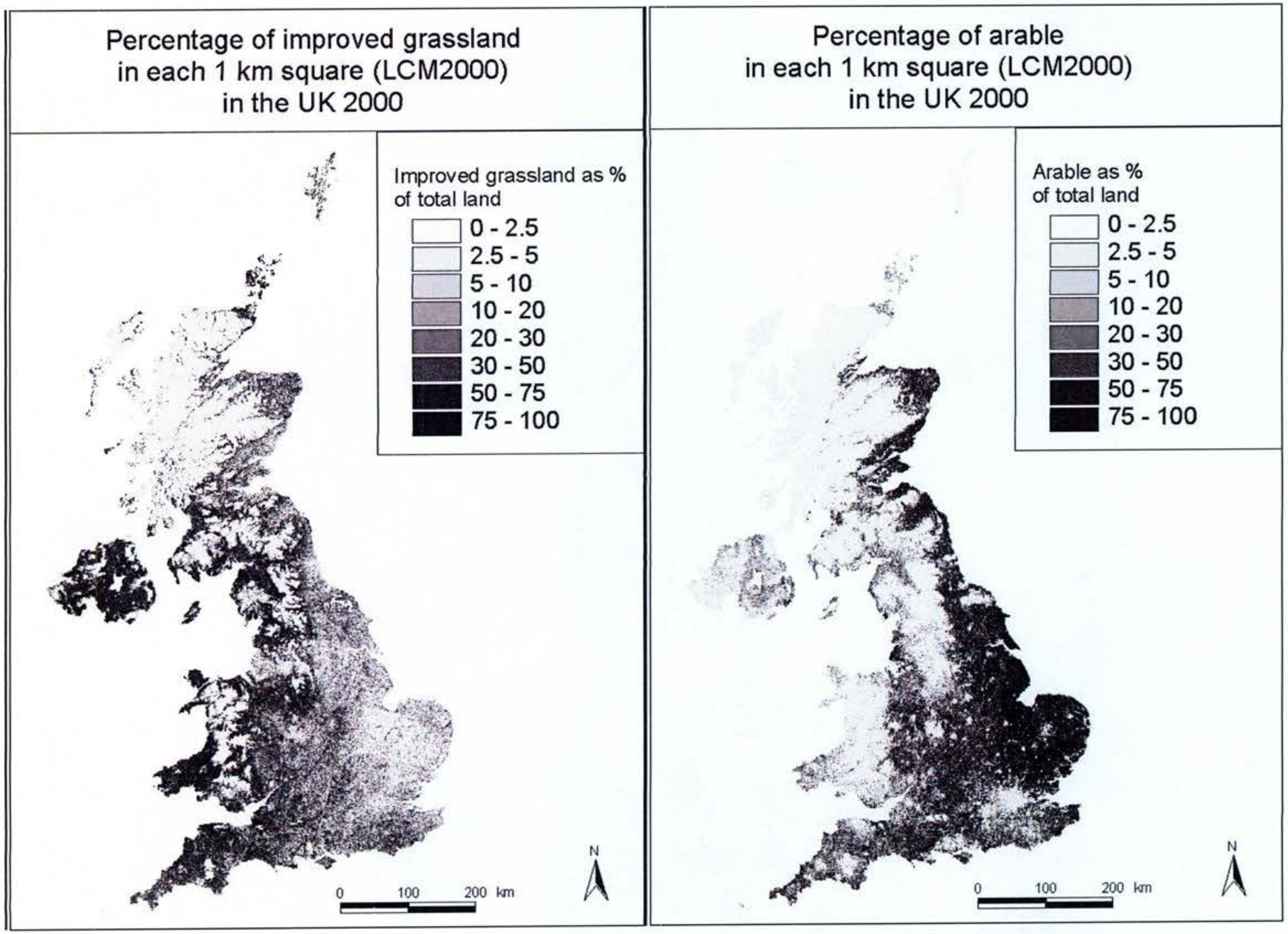

Figure 3.2.a and b: Landcover classes aggregated from the LCM2000 dataset for use in the AENEID model to spatially distribute NH3 sources, a) arable land b) improved grassland.

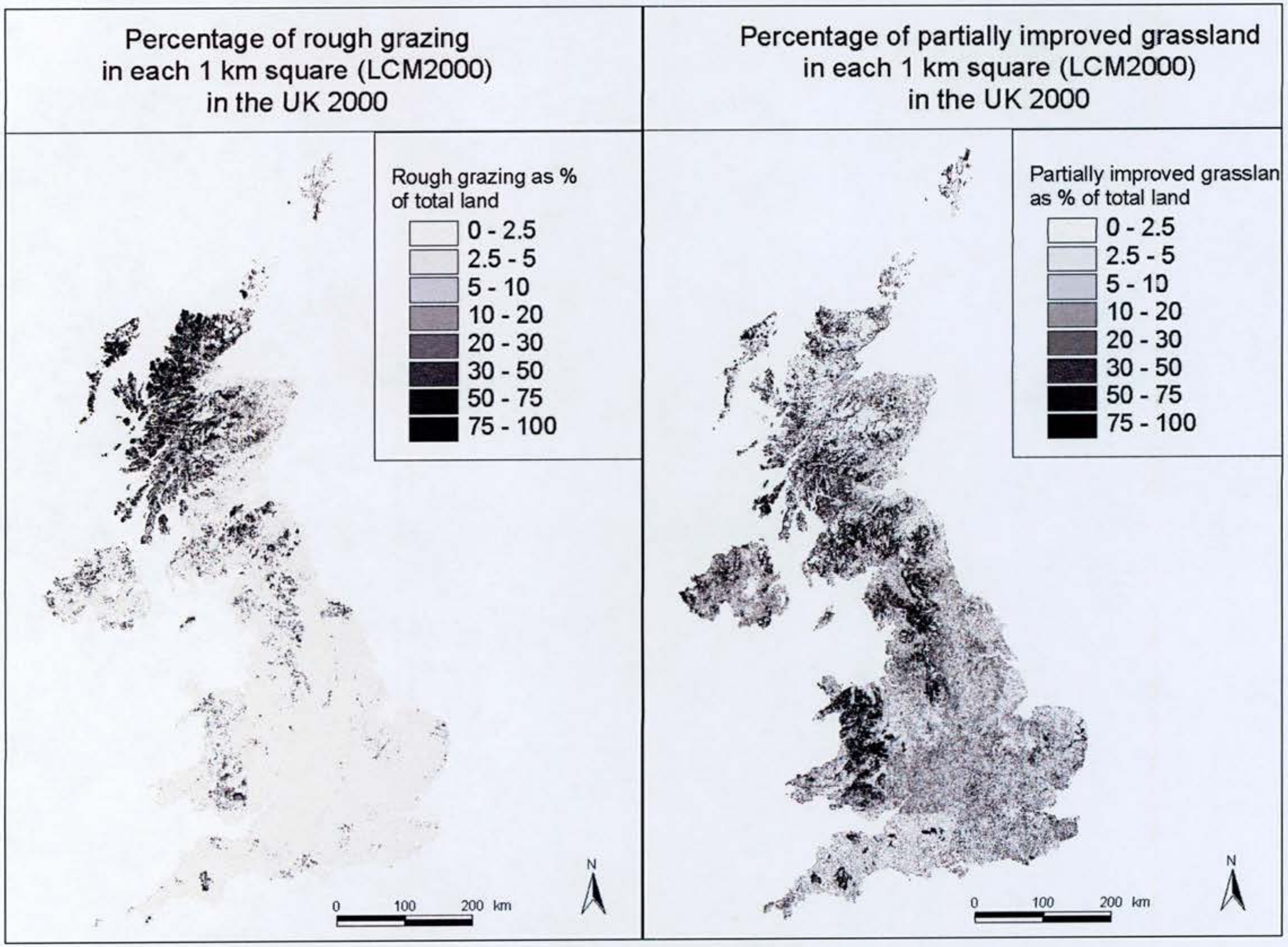

Figure 3.2.c and d: Landcover classes aggregated from the LCM2000 dataset for use in the AENEID model to spatially distribute NH3 sources, c) partially improved grassland, d) rough grazing

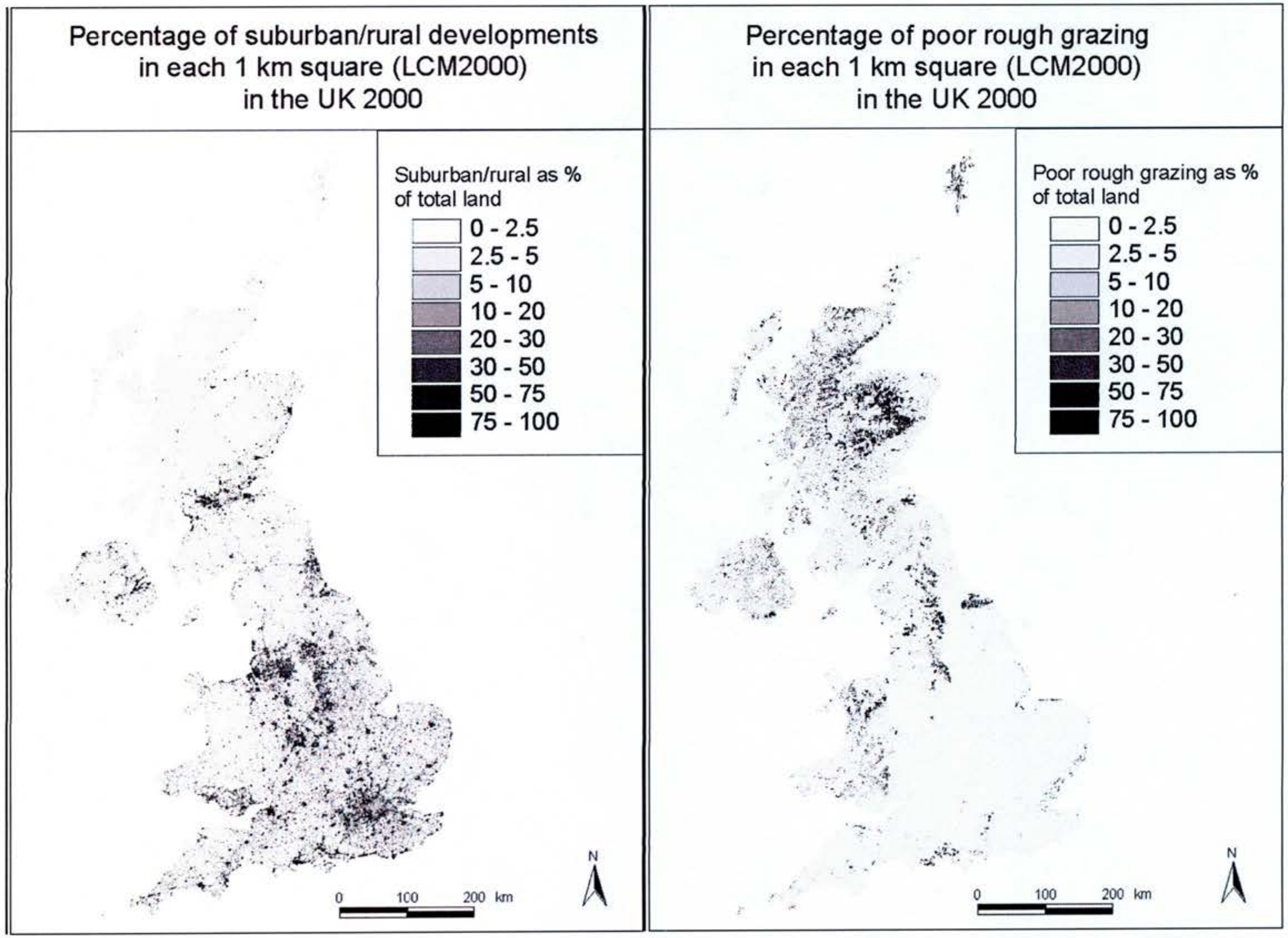

Figure 3.2.e and f: Landcover classes aggregated from the LCM2000 dataset for use in the AENEID model to spatially distribute NH3 sources, e) poor rough grazing, f) suburban/rural development.

3.4 Calculating apportioning percentages for different livestock types

The AENEID model uses different apportioning rules to allocate the source activities to various landcover classes within each aggregation zone. These apportioning rules are expressed as percentages of NH_3 emission per landcover type, and were set up to reflect the most likely landcover location of NH_3 emissions for each livestock category. *It should be stressed that the AENEID model does not aim to find the most likely location of the animals themselves on the map, but rather the most likely location of the NH_3 emission from those livestock* (Dragosits et al., 1998). Applying these rules in the model therefore made it possible to spatially distribute the livestock emissions as NH_3 sources within each aggregation zone. This requires prior information on the quantitative contribution of different agricultural activities to ammonia emissions.

The apportioning percentages were derived from the assumption that the four main livestock activities (grazing, housing, manure storage and landspreading of livestock manures) occur on specific landcover types. The apportioning is expressed as percentage values of each emission stage on the appropriate landcover types:

- housing and storage emissions occur on specific landcover, e.g improved pasture for dairy cattle, as a best approximation for the dairy farms' location within each aggregation zone;
- manure spreading emissions occur on specific landcover, e.g. arable land and improved pasture;
- grazing emissions occur on specific landcover, e.g. different quality grasslands.

Emissions from housing and manure storage were distributed jointly because it was assumed that they occur mainly in the immediate vicinity of each other. In order to separate grazing emissions onto different types of grassland, information on stocking densities for the different livestock on the various grassland types was applied (Table 3.4). Spreading emissions onto arable land and grassland were separated using information about the proportion of landspreading emissions onto arable land compared with grassland for each specific livestock derived from the IAEUK emission inventory of Misselbrook *et al.* (2004).

Table 3.4. Average annual stocking density values for grazing livestock on different grassland types and % distribution values derived for grazing animals on (grassland) landcover type. Source: Dragosits (1999), based on J. Vipond and B. Lowmon, SAC Edinburgh, pers. comm., 1996.

Livestock class	Landcover type	Avg. annual stocking density	% distribution of grazing animals
Dairy cows	Improved pasture	3 cows ha^{-1}	57.1%
	Partially improved pasture	2.25 cows ha^{-1}	42.9%
	Unfenced unimproved pasture (rough grazing)	-	-
	Total		100%
Other cattle	Improved pasture	1.75 t liveweight ha^{-1}	55.1%
	Partially improved pasture	1.375 t liveweight ha^{-1}	43.3%
	Unfenced unimproved pasture (rough grazing)	0.05 t liveweight ha^{-1}	1.6%
	Total		100%
Sheep, goats & deer	Improved pasture	10 ewes ha^{-1}	57.8%
	Partially improved pasture	5 ewes ha^{-1}	28.9%
	Unfenced unimproved pasture (rough grazing)	2 ewes ha^{-1}	11.6%
	Poor rough grazing (heather etc.)	0.33 ewes ha^{-1}	1.9%
	Total		100%

Table 3.5 shows an example of apportioning percentages for some of the livestock categories. Source activities (housing, storage, spreading and grazing) for dairy cows are distributed as a percentage of the total emission onto arable land (12 % spreading emissions), improved pasture (32 % housing emissions, 13 % storage emissions, 29 % spreading emissions and 6 % grazing emissions) and partially improved pasture (5 % spreading emissions and 4 % grazing emissions). From this distribution, the final apportioning percentages (Apportion_% in Table 3.5) are calculated per landcover category. In the example with dairy cows, the emissions are distributed onto arable (12 %), improved pasture (79 %) and partially improved pasture (9 %) within the aggregation zone.

Table 3.5. Examples of apportioning rules for some of the livestock categories.

1 - Dairy cow (milking)	Housing	Storage	Spreading	Grazing	Apportion_%
Fraction of total NH_3 emission	31.7%	12.7%	45.8%	9.8%	
Arable	(0%)	(0%)	(26%)	(0%)	
Improved pasture	(100%)	(100%)	(63%)	(57%)	
Partially improved pasture	(0%)	(0%)	(11%)	(43%)	
Apportioned fraction of total NH_3 emission					
Arable	0%	0%	12%	0%	**12%**
Improved pasture	32%	13%	29%	6%	**79%**
Partially improved pasture	0%	0%	5%	4%	**9%**
12 – Sheep	**Housing**	**Storage**	**Spreading**	**Grazing**	**Apportion_%**
Fraction of total NH_3 emission	12%	1%	10%	77%	
Improved pasture	(100%)	(100%)	(100%)	(58%)	
Partially improved pasture	(0%)	(0%)	(0%)	(29%)	
Poor grazing	(0%)	(0%)	(0%)	(12%)	
Very poor grazing	(0%)	(0%)	(0%)	(2%)	
Apportioned fraction of total NH_3 emission					
Improved pasture	12%	1%	10%	45%	**67 %**
Partially improved pasture	0%	0%	0%	22%	**22 %**
Poor grazing	0%	0%	0%	9%	**9 %**
Very poor grazing	0%	0%	0%	1%	**1 %**
17 - Pigs – boars	**Housing**	**Storage**	**Spreading**	**Grazing**	**Apportion_%**
Total	%	%	%	%	**%**
Fraction of total NH_3 emission	38%	5%	35%	22%	
Suburban	(50%)	(50%)	(0%)	(0%)	
Arable	(50%)	(50%)	(72%)	(50%)	
Improved pasture	(0%)	(0%)	(28%)	(50%)	
Apportioned fraction of total NH_3 emission					
Suburban	19%	2%	0%	0.0%	**20 %**
Arable	19%	2%	25%	11.0%	**60 %**
Improved pasture	0%	0%	10%	11.0%	**20 %**

The apportioning rules were defined for three particular groups of livestock in order that the spatial distribution of source activities could take place. These groups were a) cattle and horses, b) pigs and poultry and c) sheep, goats and deer. The apportioning percentages for each individual category within these groups were calculated, and a detailed description of the calculation of these apportioning percentages is presented below.

3.4.1 Cattle and horses

Common apportioning rules were applied for cattle and horses, because their ammonia emissions are likely to occur on similar landcover classes.

Housing and storage

Improved pasture is deemed as the most likely location for housing and storage emissions from cattle, because cattle houses, and the associated storage facilities, are likely to be situated close to good grazing land. All housing and storage emissions (100 %) were therefore assumed to occur on improved pasture.

Spreading emissions

Cattle manure is spread both onto arable land and grassland and the proportions were derived from Misselbrook *et al.* (2004). These proportions may vary for different types of livestock and are summarised in Table 3.6. Cattle are likely to graze on two different types of grassland, improved pasture and partially improved pasture, which are not distinguished in the emission inventory of Misselbrook *et al.* (2004). It was therefore assumed that 85 % of the spreading emissions to grass occur on improved pasture, and the remaining 15 % occur on partially improved pasture in accordance with the assumption made by Dragosits (1999).

Table 3.6. Proportion of ammonia emissions on arable and grassland derived from Misselbrook *et al.* (2004).

	% emissions from spreading on arable	% emissions from spreading on grassland
Cattle	25 – 30 %	70-75 %
Sheep	0 %	100 %
Pigs	61 – 72 %	28 – 39 %
Poultry	33 – 47 %	53 – 67 %

Grazing

Grazing emissions from dairy cows and other cattle vary both in magnitude and spatial location. For dairy cows, it was assumed that 57 % of the grazing emissions occur on improved pasture and 43 % on partially improved pasture. These percentages were based on average annual stocking densities according to agricultural practice (Dragosits, 1999) as shown in Table 3.4. For all other cattle, it was assumed that 55 % of the grazing emissions occur on improved pasture, 43 % on partially improved pasture and 2 % on poor grazing (Table 3.4).

3.4.2 Pigs & Poultry

Housing and storage

The locations of the poultry farms are likely to be associated with large uncertainties in the model, because there is no obvious landcover class to link them to. Poultry farms are however likely to be situated on developed ground and the landcover class 'suburban/rural development' was therefore considered as the most likely location for poultry farms. All housing and storage emissions (100 %) from poultry farms were therefore allocated to the landcover class 'suburban'.

Pig farms are also likely to be situated on developed ground and/or in close proximity to arable land, especially cereal farming, because it provides cheap bedding material, grain for feeding and also locations for the landspreading of the pig manure (D. Moorhouse, ADAS, pers. comm., 1996, in Dragosits (1999)). 50 % of the emissions from pig housing and storage were therefore assumed to occur in suburban areas and the remaining 50 % on arable land, as a best approximation.

Spreading emissions

As for cattle, spreading emissions from pigs and poultry occur both on arable land and on grassland. The proportions of emissions for the two landcover classes were derived from the IAEUK of Misselbrook *et al.* (2004) (see Table 3.6). All spreading emissions from pigs and poultry to grassland were assumed to occur on improved pasture, because the act of spreading constitutes 'improvement' (fertilizing) the grassland, and it can therefore no longer be regarded as 'rough grazing'.

Grazing

Outdoor pigs and poultry are likely to be put on arable land or grassland. It was therefore assumed that 50 % of the outdoor emissions occur on arable land, and the remaining 50 % on improved pasture.

3.4.3 Sheep, Goats and Deer

Housing and storage

As for cattle, all housing and storage emissions (100 %) were assumed to occur on improved pasture, because livestock houses and storing facilities are likely to be located in close proximity to good grassland.

Spreading emissions

All spreading emissions (100 %) were allocated to grassland in the IAEUK of Misselbrook *et al.* (2004) and all spreading emissions from sheep, goats and deer were therefore assigned to improved pasture in the model.

Grazing

It was assumed that grazing emissions from sheep, goats and deer occur on improved pasture (57.8 %), partially improved pasture (28.9 %), poor grazing (11.6 %) and very poor grazing (1.9 %). These percentages were derived from average annual stocking densities as shown in Table 3.4.

3.5 Running the AENEID model for livestock sources

In summary, the AENEID model for livestock sources shown in Figure 3.3 is calculated at a 1 x 1 km grid using the following input data:

- *Aggregation zone dataset* - A UK 1 x 1 km grid was generated with zones representing parishes for England and Scotland, grouped parishes for Wales and 5 x 5 km zones for Northern Ireland;
- *UK Agricultural Census data* – All census items were aggregated into the 46 livestock and crop categories, i.e. number of animals in each livestock

category per aggregation zone. (For Wales, dis-aggregation rather than aggregation had to be applied, see section 3.3.1);

- *Landcover data* – The proportion of each of the six relevant landcover classes for each 1 x 1 km grid square was generated, i.e. percentage of each landcover category per 1-km grid cell;
- *Apportioning percentages* – The percentage values for the distribution of each source activity to relevant landcover was calculated, based on emission data derived from the IAEUK of Misselbrook *et al.* (2004), and the stocking densities shown in Table 3.4 (Dragosits, 1999);
- *Emission potentials* – The amount of NH_3 emissions for each livestock category was derived and calculated from the IAEUK of Misselbrook *et al.* (2004), i.e. as kg NH_3-N per animal per year.

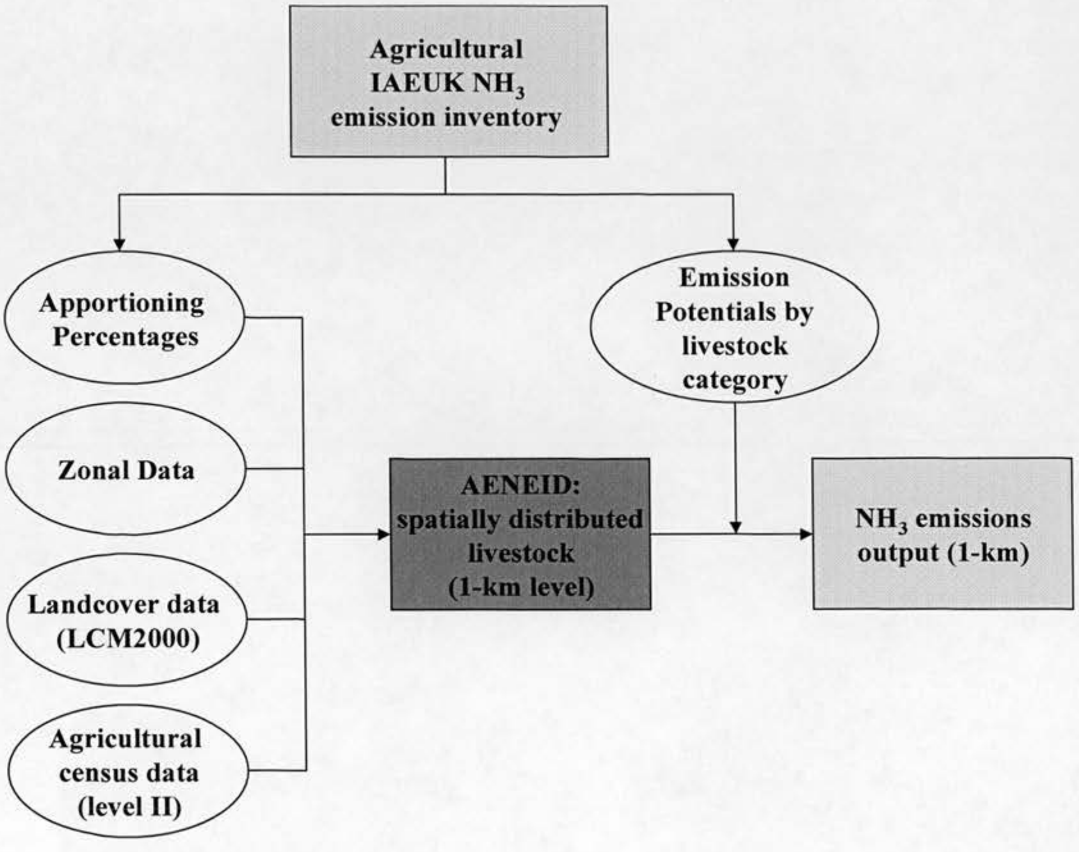

Figure 3.3. AENEID methodology to re-distribute livestock categories within the aggregation zone. Landcover data and apportioning rules were incorporated in the model to spatially distribute livestock as ammonia sources.

All the above input data, except for the emission potentials, are supplied to the AENEID model to re-distribute the livestock units at a 1 x 1 km grid within each aggregation zone. The apportioning percentages (Table 3.5) represent weightings applied to the landcover amounts in the aggregation zone, i.e. the proportion of animals assigned to a specific landcover type, e.g. arable land, is the same, independent of the amount of arable land in the aggregation zone. Consequently, in two zones with the same apportioning percentages, the emission will be more intense on arable landcover in parishes with small amounts of arable land. If a relevant landcover type is missing in a parish, the model reassigns the livestock category to the next most likely landcover type, to ensure that no emissions are "lost" in the modelling process due to discrepancies between the census and landcover data. When all livestock categories have been distributed across the UK, the estimated emission potentials for each livestock category are applied to calculate ammonia emission maps for livestock.

The simplest and most straightforward way of spatially distributing the livestock items in the aggregation zone at a 1-km grid resolution would have been to distribute them evenly within the zone. This would however have resulted in large spatial representation errors, because livestock emissions would be located in areas where they are unlikely to occur in reality, such as built up areas, lakes etc. Another option would be to distribute the livestock evenly onto agricultural land at 1-km grid resolution within the parish. This methodology is applied by the Edinburgh University Data Library (EUDL) (Hotson, 1988), but a limitation with this approach is that it does not account for the variations in NH_3 emissions between different agricultural areas and source activities, and therefore is not suitable as a basis for a spatial NH_3 emission inventory (Dragosits *et al.*, 1998). The AENEID approach on the other hand, distinguishes between intensively and extensively used agricultural areas, which further reduces the spatial representation error with regard to ammonia emissions. The model reflects higher emissions in areas of intensive agriculture and lower emissions in less intensive agricultural areas. Dragosits (1999) also showed that the AENEID approach eliminated the 'parish boundary effects', or 'border effects', clearly visible in Hotson's approach (1988) as shown in Figure 3.4.

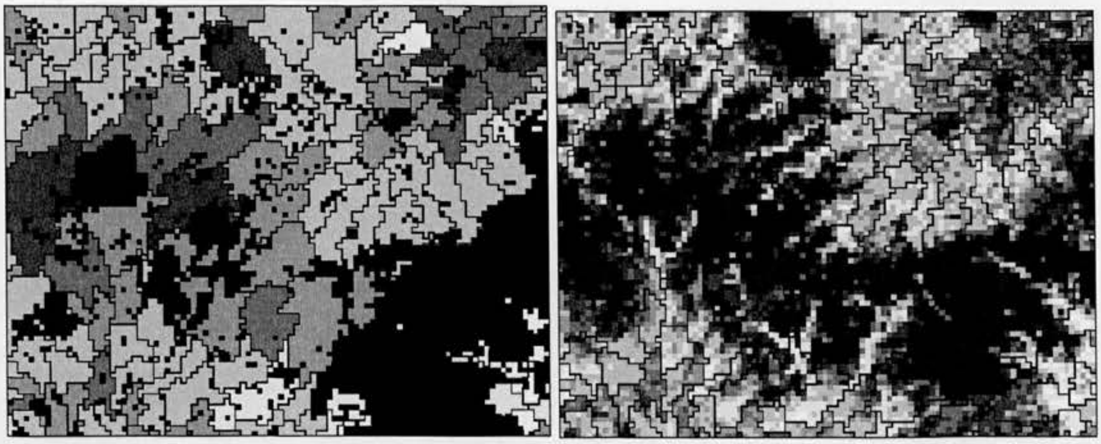

Figure 3.4. NH_3 emissions estimated at 1-km resolution for an area of the Scottish Borders. The EUDL methodology of Hotson (1988) (left) is compared with the AENEID methodology (right). White shows a high density of NH_3 emission and dark a low density of NH_3 emission. The outline areas represent parish boundaries. Source: Dragosits (1999).

3.6 The spatial distribution of ammonia emissions from fertilized crops and conserved grassland

The approach to calculate NH_3 emissions from fertilized crops and grasslands is somewhat different than for livestock. Crop and grass categories from the agricultural census provide a detailed level of arable land use, necessary to model nitrogen fertilizer use and the consequent ammonia emissions. Management practices and fertilizer application rates to different crops and types of grassland vary, and as a consequence, so does the magnitude of ammonia emission. For instance, wheat or oilseed rape receive a much larger fertilizer application rate than spring barley. Set-aside land, on the other hand, is neither grazed nor fertilized, and therefore not associated with any ammonia emissions from livestock or fertilizer sources. There may however be some small emissions during the senescence of the vegetation, or from existing stores of N due to previous fertilization or manure application, depending on whether this is short-term or long-term set-aside land.

The spatial distribution of crops and grassland as ammonia sources is a more straight-forward process than the spatial distribution of livestock. Crop statistics from the agricultural census, information on average fertilizer application rates for each crop and fertilizer emission potentials (see Table 2.11) are applied to calculate the total ammonia emission from cropland and grassland respectively for each aggregation zone. The agricultural census provides crop and grassland statistics, and the annual British Survey of Fertilizer Practice (BSFP, 2001) provides information on average N fertilizer application rates to different crop and grass categories. Total NH_3 emissions for each aggregation zone due to fertilization of crop land are distributed onto the landcover class 'arable land' in the zone, and total emissions due to N fertilization of grasslands are distributed onto 'improved grassland'.

It should be emphasised that part of the ammonia emissions from grassland have already been included with livestock grazing and manure spreading emissions. In order to avoid double counting, not all of the grass in the census was included in the model to calculate emissions from fertilizers. The grassland tables in the BSFP suggest that about two thirds of all grassland is grazed, with the remaining part being cut for hay or silage, therefore only one third of the total grass registered in the census was included in the model for the distribution of emissions from conserved grassland (Dragosits, 1999).

3.7 Calculation of ammonia emissions from non-agricultural sources

Dragosits (1999) redistributed the estimated total ammonia emission from each non-agricultural source activity onto either suitable landcover or by population census data as shown in Table 3.7. NH_3 emissions from humans, horses, pets, sewage works, transport, landfills, industrial sources, coal combustion, waste incineration and household products were scaled by population. Seabird emissions were allocated to coastal areas and emissions from wild animals were assumed to occur on natural and semi-natural land, as well as agricultural and suburban areas. Biomass burning was allocated to arable land and sugarbeet processing to individual factory locations.

Since the spatial distribution by Dragosits (1999), non-agricultural sources have been updated both regarding their emission potential and the methodology to spatially distribute some of the sources, and annual updates are now submitted to the NAEI (Dragosits *et al.*, 2004). These updates are incorporated in the new spatially distributed NH_3 emission map calculated here.

Table 3.7. Methodology to spatially distribute non-agricultural ammonia emissions for the UK by Dragosits (1999).

NH_3 Source type	Spatial allocation
Direct human emissions (breath, sweat, smoking, nappies)	Scaled by population
Horses	Scaled by population
Pets (cats & dogs)	Scaled by population
Seabirds	Coastal areas
Wild animals (deer, rabbit etc.)	Natural and semi-natural land, agricultural land, suburban areas
Biomass burning	Arable land
Sewage works	Scaled by population
Sewage sludge spreading	Agricultural land & coniferous woodland
Transport	Scaled by population
Landfill sites	Scaled by population
Industrial sources (except sugarbeet processing)	Scaled by population
Sugarbeet processing	Individual locations & emissions available
Coal combustion (domestic & industrial)	Scaled by population
Waste incineration	Scaled by population
Household products	Scaled by population

Land spreading of sewage sludge has increased following the 1999 ban on dumping of sewage sludge at sea. Ammonia emissions from horses and wild animals have also increased because the estimation of the horse population and number of wild animals (especially red deer) have been revised (Dragosits *et al.*, 2004). The methodology to spatially distribute NH_3 emissions from horses has been revised in order to smooth out the unrealistic emission peaks in the most densely populated urban areas as a result of mapping these emissions by human population. The approach to spatially distribute these sources therefore assumed that fewer horses per population were located in densely populated urban areas.

Seabird emissions have been updated both regarding their emission potential and their spatial distribution. The emission potential of seabirds has been reduced, following detailed research by Blackall (2004) and Wilson *et al.* (2004). Wilson *et al.* (2004) also developed a new approach to spatially distribute seabird emissions, by mapping the JNCC seabird count at 1-km grid resolution. This dataset is based on counts collected by observing bird colonies, mainly from segments of coastal areas and sea cliffs.

Landfill and sewage works emissions were updated in the National Atmospheric Emission Inventory (NAEI) (Dragosits *et al.*, 2004), applying data from the

Environment Agency (EA) website for England and Wales and from the Scottish Environmental Protection Agency (SEPA) for Scotland. The location and size of landfill sites were available for Scotland. For England and Wales, the location and a classification by size (large/medium/small) was available for a number of sites. After distributing the landfill sites for the NAEI (2004), the density of English and Welsh landfill sites was however much lower than for Scotland, and accounted for a much lower per-person equivalent of the known waste put into landfill sites. This gap was filled by distributing the shortfall (31 %) via the population census, but excluding densely populated urban areas (> 1000 people per square kilometre) and taking account of the distribution of the known landfill sites. For Northern Ireland, the emissions were mapped by population, because no information of the location of landfills were available. This approach has the advantage of spatially locating the landfill emissions were they are likely to be, rather than scaling them by population and generating large emission peaks in urban centres.

3.8 Sources of uncertainty in the AENEID model

Modelling ammonia emissions involves converting various input data into spatially distributed ammonia emissions using mathematical expressions. Uncertainties in the modelling result are therefore associated with the quality of the input data, as well as the mathematical expressions applied in the model. In a modelling context it is commonly accepted that the quality of the (input) data strongly affects the modelling result, and that uncertainties in the input data will generate uncertainties in the final result. Understanding the limitations of the (input) data is therefore vital. The more detailed the level of modelling, the more significant does the quality of the input data become, as aggregated information (aggregation of large geographic areas and/or broad activity types) tends to smooth out extremes caused by some of the uncertainties. Aggregated data have, on the other hand, the disadvantage of a loss of detail, so too much smoothing may also have a detrimental effect on the modelling result. In this section both uncertainties in the input data and the modelling process of the AENEID model are discussed, with a focus on the original AENEID model, and solutions are proposed on how some of these uncertainties can be reduced in the new AENEID model.

3.8.1 Uncertainties in the agricultural census data

As discussed previously, the agricultural census data are the most important input to the model. Uncertainties in the dataset will therefore have significant implications on the modelling result. Uncertainties in the census may be of two types, either due to statistical errors or associated with the location of the farm. Statistical errors are associated with the quality of the information collected, and/or the methods used to compensate for non-respondents. Some farms may not have been included in the census, either because they are not known to exist, or because they are small (minor holdings) and therefore only surveyed periodically. Data may also be missing due to a low response rate. Although farmers are legally obliged to complete the census, they do not always comply. In Wales, for example, the response rate is only about 75 %, and the Welsh Assembly (responsible for collecting the data) also see a trend of a low response rate being more common among large farms (S. Neil, The Welsh Assembly, pers. comm. 2003). This is worrying, because large farms contribute a greater proportion of NH_3 emissions than smaller farms. In order to compensate for non-respondents, values may be estimated, i.e. new values are calculated based on previous years and the likely change reflected by other respondents. When estimated values are based on a chain of previous estimations, the values are highly uncertain. These types of uncertainties are very difficult to quantify and the level of error may even vary between the devolved regions, as the methods of collecting data and compensation for non-respondents may be different. The Welsh Assembly has chosen to aggregate some of the agricultural census categories (see Section 3.3.1) to smooth out some errors, but this is at the expense of detail.

Another major uncertainty in the agricultural census is the location of the farm. The geographical reference of the farm in the census may only be an approximation of the actual location of the farm. In addition to this, a major problem is that there is no relationship between the boundaries of the farm and the aggregation zone. Therefore, even if the exact location of the farm building was known, the area of the farm may only partially be located in the aggregation zone where it is recorded and overlap into one or more other zones (Hotson, 1988). For some of the major intensive pig and poultry farms, where many livestock units are represented, it may be that the location of the 'farms' reported in the agricultural census is actually the location of the head office, and not the location of the farms themselves.

Figure 3.5 shows a comprehensive overview of some of the errors that may arise as a consequence of uncertainties associated with the location of the farm. Due to these errors, farmland is likely to be over- or underestimated, especially in small aggregation zones. The agricultural area reported in the census for all farms recorded in the aggregation zone may be greater than the actual area of the aggregation zone. These uncertainties due to the aggregation of census data into larger entities (the Modifiable Areal Unit Problem, MAUP) are further discussed in Chapter 9.

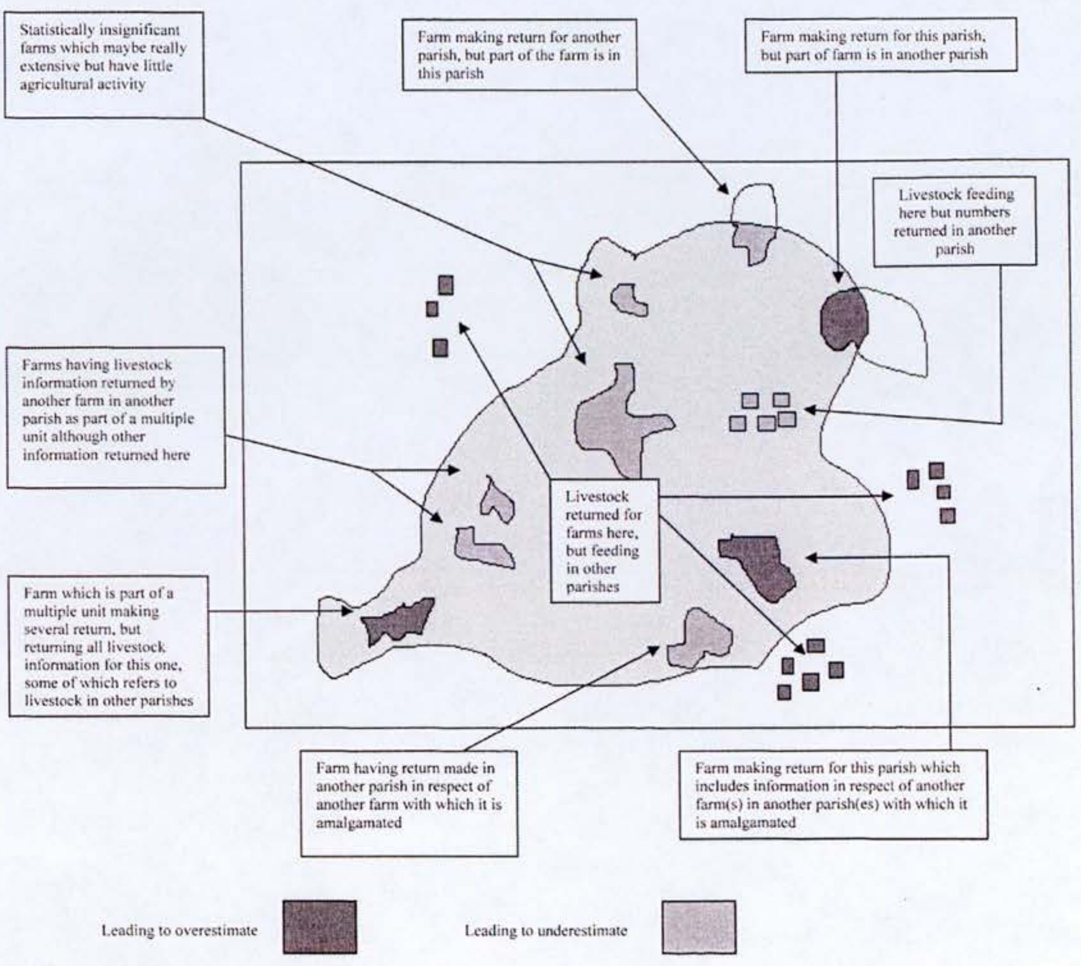

Figure 3.5. Uncertainties in the agricultural census associated with the spatial location of the farm and its agricultural land. Source: Hotson (1988).

3.8.2 Uncertainties in the landcover data

Both the landcover dataset from 1990 (LCMGB) and the landcover dataset from 2000 (LCM2000) were derived from the classification of satellite images. The accuracy of the classified image depends on many different factors, such as time of year, classification method applied and interference by the atmosphere (Campbell, 1996). Fuller *et al.* (2002) have estimated the accuracy of LCM2000 to about 85 % compared with an accuracy of about 80 % for LCMGB from 1990 (Fuller et al 1998).

3.8.3 Uncertainties in the emission potentials

Calculating NH_3 emission potentials for agricultural sources is a complicated process with many uncertainties, as discussed in Chapter 2. In the original AENEID approach the source strength applied was an average value for the whole country, but the emission potentials may vary locally due to environmental factors and farming practice. A good example of this variability of the emission potential is the grazing emission from cattle. The grazing season varies widely throughout the UK and is generally longer in the south than in the north, due to climatic conditions (Gregory, 1964). At present the model does not account for this variability, but applies average values. This is likely to have a significant impact on the spatial distribution of emissions, because grazing animals are associated with considerably lower overall emission rates than when cattle are housed.

Uncertainties due to variable farming practice are more difficult to estimate, because of the lack of management data for individual farms. These data are difficult to obtain, and if they were included in the agricultural census questionnaire it is likely that the number of non-respondents would increase significantly because of the extra effort required by the farmer. The new IPPC directive, targeting the pig and poultry industry (see Section 1.4.4) may however improve the accessibility of specific farm data on large pig and poultry farms (SAC, 2001). This would be a great advantage, because uncertainties associated with farming practice are likely to have a greater significance for large intensive farms than for smaller farms.

3.8.4 Uncertainties in the spatial distribution of emission sources

Not only the input data, but also the methodology for distributing the emission sources is associated with uncertainties. The AENEID distribution methodology assumes average farming practice and does not take into account regional variations that might occur, e.g. a higher proportion of manure may be spread on grassland in areas dominated by grassland with a limited area of arable land.

Other uncertainties in the AENEID methodology are due to the assumption that all emissions occur in the aggregation zone of origin (see Section 3.8.1). Although this assumption may be acceptable in most situations, especially for land-based farming of cattle and sheep, for emissions from intensive pig and poultry farms the spatial uncertainties are much larger. Pig and poultry manures are often transported over long distances for use on other farms, and poultry manure may be incinerated. Ammonia emissions from spreading of pig and poultry manure are therefore likely to be overestimated within the zones (parishes) of origin of these farms, according to the current methodology.

3.8.5 Temporal uncertainties

The AENEID model developed by Dragosits (1999) was designed for annually averaged values. In reality NH_3 emissions have a high temporal variability and vary with hours, days, months, seasons and between years. These temporal variations are due to meteorological conditions as well as the seasonal trends in agricultural practice (see Section 2.3 and 2.4). Some agricultural activities only occur during a part of the year, such as grazing, and strongly affect both the magnitude and the spatial location of emissions. Emissions from grazing occur during the whole grazing season at varying intensity, while other activities, such as manure spreading, give rise to very high but short-term emission. Due to the temporal variability in NH_3 emissions during a year, it is therefore desirable to develop a model that can integrate temporal aspects of NH_3 emissions.

3.9 Main areas for improvement in the AENEID model

Based on the sources of uncertainty described above, three main weaknesses in the original AENEID model have been identified:

1) *Ammonia emission potentials vary across the UK, due to the diversity in environmental factors and agricultural practices.*
2) *Ammonia emission potentials vary during the year.*
3) *A proportion of the ammonia emissions from large intensive pig and poultry farms is likely to occur outside the aggregation zones of origin.*

3.9.1 Applying regionally varying emission potentials

Chapter 2 discussed the diversity in environmental factors and agricultural practices that affects the variability of NH_3 emission potentials in the UK. Developing a model that takes these variations into account could be achieved by incorporating factors that influence the spatial variability of NH_3 source strength data. These factors can be either environmental factors (such as temperature variations influencing the length of the grass growing season, and therefore the grazing season), or the influence of agricultural practice, such as regional variations in fertilizer application rates. Recently, a system that incorporates some of these factors has been developed within *the National Ammonia Reduction Strategy Evaluation System* (NARSES, see also Section 1.5.5 and Section 4.2). In Chapter 4 an approach is developed, which introduces spatially varying emission potentials by coupling AENEID with the NARSES model.

3.9.2 Applying temporal emission potentials

The strong seasonal fluctuation of ammonia emission during a year justifies the application of a higher temporal resolution than annual emissions. Calculating seasonal or monthly emission maps could take into account some of the temporal variability and therefore improve the calculated emission result. The development of a monthly emission model and the spatial and temporal dis-aggregation involved are further discussed in Chapter 5.

3.9.3 A sub-model allowing manure removal from the zone of origin

The original AENEID model operated at a parish level and assumed that all emissions from animals within the parish occurred in the parish of origin. As already mentioned, manure may be transported between different parishes, especially in parishes with large intensive pig and poultry farms. In addition to this, a significant proportion of broiler manure in the UK is incinerated in special powerstations and should thus be excluded entirely from landspreading. It is therefore desirable to develop a sub-model for manure movement between aggregation zones and to compensate for the removal of emissions due to incineration in a spatially distributed approach rather than by reducing emissions by equal amounts across the UK. These issues are further discussed in Chapter 7.

3.10 Further development of the AENEID model

The new AENEID model presented in this thesis is the result of modifications to the original model developed by Dragosits (1999). The new model has taken on board concerns over the uncertainties discussed in the previous sections, and methods were developed to improve the spatial distribution. These include applying variable emission potentials, calculating monthly emission maps to take into account intra-annual variation, and developing a sub-model to compensate for pig and poultry manure removal from the aggregation zones of origin. In addition to these three main enhancements to the model, other areas of development include the following:

- Enlarging the spatial extent of the model;
- Improving the quality of the input data;
- Developing the modelling methodology;
- Improvements to non-agricultural sources (not as part of this thesis);

3.10.1 Enlarging the spatial extent of the model

The spatial extent of the ammonia emission inventory has been increased in the new model compared with the original AENEID model. In the new approach, in addition to the UK, the Isle of Man and the Scilly Isles are also included.

3.10.2 Improving the quality of input data

The new AENEID model uses updated landcover data from 2000 (LCM2000), where all areas have been classified, whereas the older landcover dataset applied in the previous inventory contained unclassified areas mainly due to cloud cover. In addition, the spatial extent of the newer dataset is greater, also including Northern Ireland, and not only Great Britain.

The IAEUK is being updated annually, and the latest available research findings and knowledge of agricultural practice (Misselbrook *et al.*, 2004) were implemented here. The updated emission potentials (see Section 2.5.1) are applied in the new model to calculate the apportioning percentages (see Section 3.4) and the total NH_3 emission.

The new model has a higher level of detail regarding the number of livestock categories applied. The old model was designed for 12 livestock categories, whereas the new model uses 31 categories (Table 3.8). This minimises the uncertainty associated with intra-category variation in source strength and has mainly been possible due to a significant improvement in computing facilities over the last decade or so.

Table 3.8. Comparison of number of livestock categories applied in the original AENEID model compared with the new model.

Original AENEID model **- 12 livestock categories**	**New AENEID model** **- 31 livestock categories**
Dairy cows	3 categories
Other cattle	8 categories
Sheep	1 category
Lambs	1 category
Sows	3 categories
Fatteners and others	6 categories
Layers	1 category
Broilers	1 category
Other poultry	4 categories
Horses	1 category
Goats	1 category
Deer	1 category

3.10.3 Developing the modelling methodology

The model code has been modified to allow greater flexibility, so that it can easily be applied to different years, as well as incorporating temporal, and/or spatially varying emission potentials and apportioning percentages. The original model applied static input constants, whereas the new version applies input variables either via a separate parameter file, or as grids, thereby allowing more flexibility to handle spatially variable apportioning percentages and emission potentials. This coupled with the ability to vary the temporal resolution of the calculations allows for temporal as well as regionally varying emission potentials.

The modelling methodology has also been improved regarding land spreading emissions of manure. The apportioning percentages are now based on Misselbrook *et al.* (2004) calculations of emissions from manure and slurry to arable and grassland respectively for each individual category, instead of an average value for the whole livestock sector as was applied in the past.

3.10.4 Improvements to non-agricultural sources

Non-agricultural sources were updated both regarding their emission potential and the approach to spatially distributing them. These updates integrate a number of different studies which attempted to address some of the limitations in the non-agricultural NH_3 estimates in the original AENEID model and were described in detail in Section 3.7.

3.11 Summary and conclusions

The AENEID model (Atmospheric Emissions for National Environmental Impacts Determination) is a well established tool for fine resolution spatial distribution of NH_3 emissions in the UK. The model calculates emission maps in two steps. Firstly, it spatially distributes the emission sources and secondly, it assigns an emission potential to these emission sources. These NH_3 emission sources can be either agricultural, such as livestock and grassland, or non-agricultural, such as humans, wild animals or industries.

Agricultural statistics are one of the most important inputs to a spatial NH_3 emission inventory. These data are obtained as zonal parish aggregates and therefore have to be re-distributed over the parish zone using landcover data. Apportioning percentages, describing onto which landcover the source categories within each parish zone should be assigned, are calculated based on the agricultural NH_3 emission inventory (IAEUK) by Misselbrook *et al.* (2004). When all source activities (grazing, housing, storage and landspreading) have been spatially distributed, an emission potential, also derived from Misselbrook *et al.* (2004), is applied to calculate the final emission map.

The total ammonia emission from fertilizers is calculated for crops and grassland respectively for each aggregation zone. The calculation is based on crop statistics (derived from the agricultural census), average fertilizer application rates for each crop (derived from the British Survey of Fertilizer Practice, BSFP) and emission potentials (derived from Misselbrook et al. (2004)). Finally the NH_3 emission for each aggregation zone is distributed within the zone onto crop land and grassland, respectively.

The methodology to calculate the spatially distributed NH_3 emissions from non-agricultural sources in the AENEID model is more straight-forward, but more diverse than for agricultural sources. Ammonia emissions from non-agricultural sources (humans, horses, pets, wild animals, sewage works, transport, landfills, industrial sources, biomass burning, coal combustion, waste incineration and household products) were distributed onto either suitable landcover or by population census data.

Some of the main weaknesses in the AENEID model include the application of average annual emission potentials, and the assumption that all emissions from livestock occur in the aggregation zone of origin. These uncertainties justify the need for a sub-model allowing for manure removal from the aggregation zone of origin (particularly pig and poultry manure), as well as a spatial process-based model of emission source strength and estimates of the intra-annual variability of NH_3 emissions. All of these uncertainties have been addressed in the new AENEID model developed in this thesis (see Chapters 4, 5, 6 and 7).

4 Coupling NARSES and AENEID with the aim to derive emissions with variable source strength

4.1 Introduction

One of the key uncertainties identified in the original AENEID model is that it applies average emission potentials across the whole country. As described in Section 2.3 and 2.4, the UK is characterised by a huge diversity in environmental factors and farming practices. Applying spatially varying emission potentials that account for some of this variability is therefore expected to improve the modelling result. For instance, additional data sources, such as soil quality data, could allow the model to determine whether manure can be incorporated into the soil or not, which would be useful for investigating the applicability of proposed abatement measures and their effects on total NH_3 emissions. Incorporation of additional data to account for some of the spatial variability in emission potentials has already been achieved in the *'NARSES N-flow model'*, a software application developed within the NARSES (National Ammonia Reduction Strategy Evaluation System) project at a 10 km grid resolution.

The *NARSES N-flow model* calculates NH_3 emissions by combining data of livestock numbers with an integrated landcover/landuse dataset and regional and national values derived from farm management surveys. The emission calculation for livestock excreta is based on the Mass-Flow approach, where N flows through the manure management stages with losses occurring at each stage, expressed as a proportion of the N available at the previous stage (NARSES, 2004). The NARSES system contains spatial data that may affect the emission potential locally, e.g. soil information, hence is capable of incorporating spatially varying factors affecting the emission potential of NH_3. In the present study, the emission outputs of the NARSES N-flow model were therefore linked with the AENEID model (the coupled NARSES-AENEID model) to incorporate regional variations in emission potentials in the fine scale modelling approach of AENEID.

In this chapter, ammonia emission results of the original AENEID approach (based on the Unit-approach) are compared with the NARSES-AENEID approach, noting the effects of using the different aggregation of agricultural statistics in the form of either a) non-disclosive data (as used in NARSES) and b) disclosive input data (available only for AENEID). As has been explained in Section 3.3.1 in detail, despite using disclosive *input* data in the AENEID model, the spatial redistribution approach and subsequent combination of source categories ensures that the *output* is non-disclosive.

A key target of the NARSES project was to develop a PC-based application to estimate the spatial distribution of NH_3 emissions in order to predict abatement potential and abatement cost curves for use by both scientists and policy analysts. Central challenges have been to enable the use of spatially varying emission source strength estimates and abatement, rather than the previous more general approach using constant emission potentials for each source.

The NARSES project aimed at providing the emission maps in a form that is openly shareable, and to deliver a tool that can link to other applications (e.g. AENEID, atmospheric transport modelling and environmental effects assessment). In meeting these objectives, a NARSES desktop application has been developed by the consortium with the software development undertaken at ADAS, Wolverhampton, using a 'non-disclosive' spatial agricultural census dataset (10 x 10 km grid resolution). The input and output data of the NARSES application are in this way ensured to be non-disclosive, which is necessary for the sharing of this system with users, and allows detailed analysis of individual census categories as well as to address spatial variability at a scale suited to costs and abatement. However, the 10 x 10 km spatial resolution of the NARSES system does not match the fine scale modelling approach (1 x 1 km) of the AENEID model, where zonally aggregated confidential agricultural census data are linked with fine-scale land cover data to improve the resolution of the NH_3 emission maps, following the methodology described in Chapter 3. However, linking the two models into a coupled NARSES-AENEID model, the strengths of the NARSES N-module (variable emission source strength) can be combined with the strength of the AENEID model (fine resolution modelling).

4.2 The NARSES N-flow module

4.2.1 NARSES input data

Spatial data used within the NARSES N-flow model include (NARSES, 2004):

- Soil data for England and Wales at 1-km grid resolution (from the National Soil Resources Institute (NSRI) SEISMIC database of soil associations).
- Meteorological data at 10-km grid resolution (from the UK Climate Impacts Programme, UKCIP).
- The Land Cover Map 2000 (LCM2000) at 1-km grid resolution (Fuller *et al.*, 2002).
- Non-disclosive agricultural census data for England, Scotland and Wales (from Edinburgh Data Library) and for Northern Ireland (from DARD, Department of Agriculture and Rural Development in Northern Ireland) at 10-km grid resolution.
- Information on agricultural practice, mainly derived from farm and manure management surveys across England and Wales.

4.2.2 NARSES methodology

The NARSES system combines the data of livestock numbers with the regional and national values in the spatial input datasets described above, to calculate ammonia emissions at 10-km grid resolution (Figure 4.1). The emission calculation for livestock excreta is based on the Mass-Flow approach (see Section 1.5.1), where NH_3 may be successively lost through volatilization at any stage of manure management from a pool of total ammoniacal-N (TAN) (NARSES, 2004). During the chain of manure management stages, no newly generated TAN is added to the flow, and finally the TAN is absorbed onto soil colloids following deposition or manure application to land (Figure 4.2). This approach has the advantage of allowing calculation of the consequences of NH_3 emission reductions at one stage of manure management (upstream) on emissions at later stages of manure management (downstream).

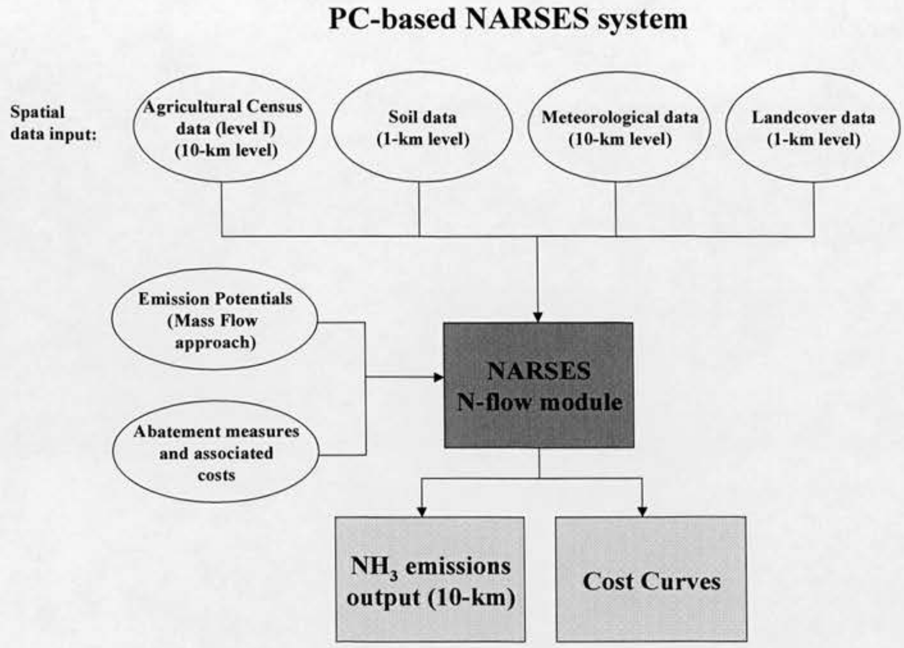

Figure 4.1. NARSES system methodology for calculating the national spatial distribution of NH_3 emissions at a 10-km grid resolution. The emission potentials are expressed as % of TAN and are based on a Mass-flow approach.

The calculation of fertilizer emissions within the NARSES system will be based on process based models that take into account the variation in climate, soil type and management practice across the UK to generate emission factors for different N fertilization types (NARSES, 2004). The fertilization model only gives the short-term emissions (within 2 - 3 weeks after fertilization application) and therefore fails to consider other crop emissions, e.g. when grass is cut. Currently the fertilization model within NARSES is not fully developed.

4.3 The coupled NARSES-AENEID approach

The coupled NARSES-AENEID approach was developed as a means to link the two models (NARSES and AENEID) to take advantage of the respective strengths of the models (Section 4.1). It has therefore been a central challenge in this thesis to develop an approach that links:

1) the spatially variable emission and abatement potentials of the NARSES module
2) the disclosive fine scale livestock and crop information (input to AENEID)
3) the effects of fine-scale differences in land cover in the AENEID model

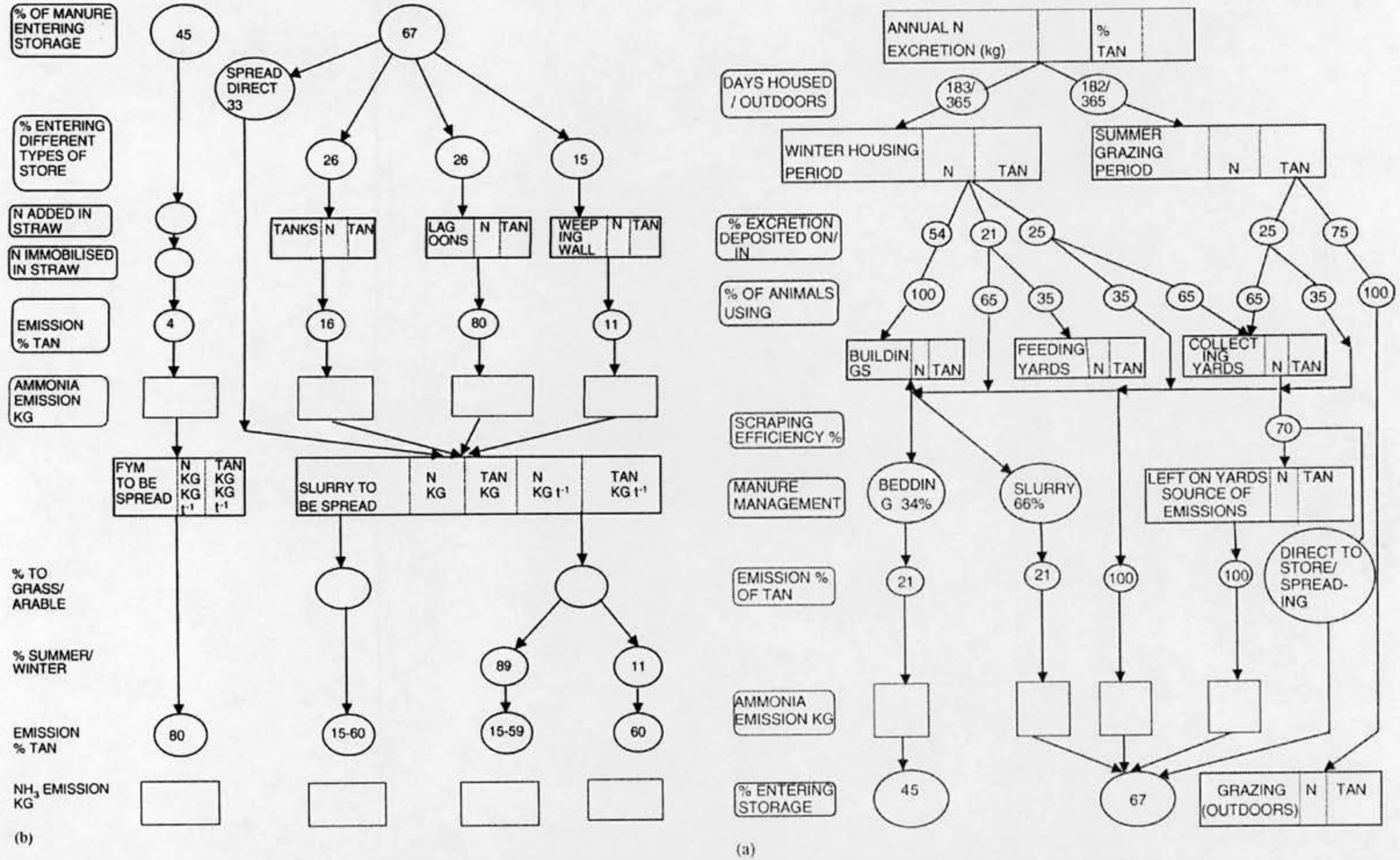

Figure 4.2. Example of the flow of TAN and the NH_3 volatilisation rates (expressed as percentages) through the chain of animal husbandry stages. Source: Webb and Misselbrook (2004).

The link between the two models is represented by spatially distributed source strength estimates from the NARSES N-flow model which are used as input to the AENEID model.

4.3.1 NARSES-AENEID data input

As discussed previously, NARSES uses non-disclosive agricultural data, while AENEID uses disclosive input data. Agricultural census data as input to the coupled NARSES-AENEID approach are therefore provided at two levels:

- **Level-I** – for use in the NARSES N-flow module (resolution: 10 x 10 km) – *non-disclosive data derived from Edinburgh University Data Library (EUDL).*
- **Level-II** – for use in the AENEID high resolution modelling (resolution: the aggregation zones) – *disclosive data acquired from the devolved regions within the UK.*

Data for three different years (1990, 1996 and 2000) provided from six different data sources were used, and all data sets had to be processed into a common format before use in the coupled NARSES-AENEID system, as well as in the NARSES N-flow module system. Data providers and data formats for the different regions within the UK for Level-I and Level-II agricultural census data are summarised in Table 4.1.

Table 4.1. Summary of data providers and data formats for Level-I (the NARSES N-flow module) and Level-II (the AENEID high resolution modelling).

Data Provider	Area	NARSES Level-I (10 km)	AENEID Level-II (1 km)	Data format
EUDL	England	X		2 x 2 km grid
	Wales	X		2 x 2 km grid
	Scotland	X		2 x 2 km grid
DARD	Northern Ireland	X	X	Rural district level (1990) 5-km grid (1996 & 2000)
Defra	England		X	Holding level
Welsh Assembly Government	Wales		X	Grouped parish level
Scottish Executive	Scotland		X	Parish level
Isle of Man Government	Isle of Man	X	X	Country level

Non-disclosive Level-I data for England, Scotland and Wales were provided by EUDL at a 2 x 2 km resolution for application in the NARSES N-flow module. 1990 data were not available, and 1988 data were used instead, which were scaled to reflect the regional statistics for 1990.

Disclosive Level-II data for England, Scotland and Wales for application in the AENEID model were provided by the devolved regions at different resolutions, as described in Chapter 3. The Northern Irish datasets were provided at 5 x 5 km grid resolution for 1996 and 2000 and rural district level for 1990, which were processed as inputs to both the NARSES PC application (Level-I) and the AENEID model (Level-II). The Isle of Man Government provided agricultural census statistics for the island as a whole, which were used as inputs to both NARSES (Level-I) and AENEID (Level-II).

All datasets had to be modified to a common format, either the 'NARSES format', or the 'AENEID format', before they could be used as input to the spatial ammonia emission inventory. This included aggregating all datasets into the 46 NARSES categories (Table 4.2), re-sampling the datasets to the required spatial resolution, i.e. 10 x 10 km for Level-I, and projecting the Northern Irish data to the Great Britain Ordnance Survey Grid.

Common categories

While it was straightforward to generate the 46 categories for Level-II data by aggregating the census items into the 46 categories, it proved to be a more complicated process for Level-I data, because some census categories were missing or amalgamated into larger groups for confidentiality purposes. It was therefore necessary to use other statistical sources to fill the data gaps for Level-I. A similar approach was applied as for the Welsh Level-II data (as described in 3.3.1), i.e. scaling the total value of each missing category derived from the National Assembly for Wales (WA, 2001) to the corresponding category for the specific region.

Table 4.2. 46 NARSES categories (31 livestock categories and 15 arable and grassland categories) applied in the NARSES system and in the AENEID model.

	NARSES-Id	Description
Cattle	1	Dairy cows & heifers
	2	Dairy heifers in calf, 2 years and over
	3	Dairy heifers in calf, less than 2 years
	4	Beef cows & heifers
	5	Beef heifers in calf, 2yrs and over
	6	Beef heifers in calf, less than 2 years
	7	Bulls >2yrs
	8	Bulls 1-2yrs
	9	Other cattle, over 2yrs
	10	Other cattle, 1-2yrs
	11	Other cattle, under 1yr
Sheep	12	Sheep
	13	Lambs, under 1 year old
Pigs	14	Sows in pig & other sows
	15	Gilts in pig & barren sows for fattening
	16	Gilts > 50kg not yet in pig
	17	Boars
	18	Other pigs, 110kg and over
	19	Other pigs, 80-110kg
	20	Other pigs, 50-80kg
	21	Other pigs, 20-50kg
	22	Other pigs, under 20kg
Poultry	23	Layers
	24	Breeding birds
	25	Broilers
	26	Pullets
	27	Turkeys
	28	Other poultry
Horses	29	Horses
Goats	30	Goats
Deer	31	Deer
Arable Land	32	Set-aside land
	33	Wheat
	34	Winter Barley
	35	Spring Barley
	36	Oilseed rape
	37	Potatoes
	38	Other cereals
	39	Sugar beet
	40	Other root crops
	41	Other crops
	42	Vegetables for human consumption
	43	Fruit
	44	Bulbs, flowers and nursery stock
Grassland	45	Grassland under 5 years
	46	Permanent grassland

The calculation may be illustrated by reference to pigs in England and Wales in 1996. In the data provided by EUDL, three NARSES pig categories (liveweights of 20 – 50 kg , 50 – 80 kg, 80 – 110 kg) were amalgamated into one category of 'other pigs' (20-110 kg). By using statistical data aggregated for the whole countries of England and Wales, but with more detailed pig categories (GSS, 1997), the non-disclosive census data were split into the three pig categories, by scaling with the aggregated category populations for each country. Although application of the data in this way does not improve the data quality, it enables the overall data categories to be used in a consistent format between years, regions and between Level I and II.

Resampling

Resampling into a common grid resolution (10 x 10 km) was only necessary for Level-I data, which was provided at 2 x 2 km (Great Britain) and 5 x 5 km (Northern Ireland) resolutions. The Level-II agricultural census data did not have to be re-sampled, as it was redistributed at a 1-km grid within the AENEID model.

The Northern Ireland data for 1990 at rural district level were spatially distributed within the rural districts through integration of land cover data (Fuller *et al.*, 2002) following the AENEID approach described earlier in Chapter 3, before they could be re-sampled to a 10 x 10 km grid resolution. This methodology was also applied for the Isle of Man. In addition all data from Northern Ireland were re-projected from the Irish OS Grid to the GB OS Grid in order to allow modelling on one common grid system.

4.3.2 The coupled NARSES-AENEID methodology

The original AENEID model described in Chapter 3 constitutes the basis for the coupled NARSES-AENEID approach applied here. The emission output from the NARSES N-flow module (10-km grid resolution) (Figure 4.1) is integrated with agricultural statistics (at 10 x 10 km resolution) to derive emission potentials for each grid cell, which can be used as spatially variable emission potentials in AENEID instead of the average values derived from the IAEUK (Figure 4.3). These regional emission potentials were also used as the basis for calculating regional apportioning percentages applied in AENEID. An overview of the data flow in the coupled NARSES-AENEID model is shown in Figure 4.3, where the link between the two models is represented by the 10-km NH_3 emission potentials derived from the NARSES system.

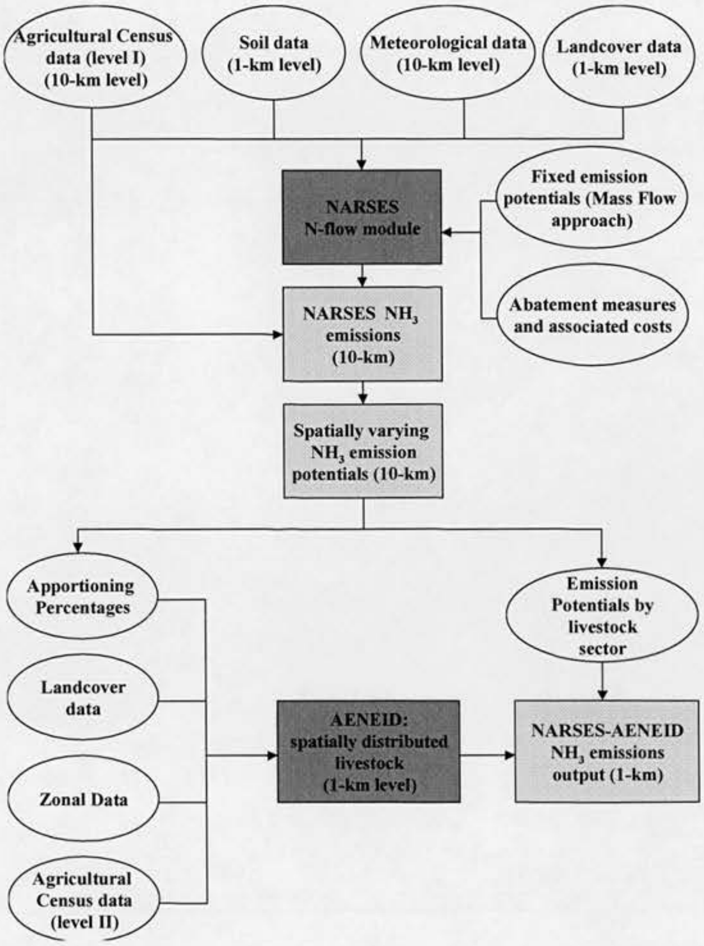

Figure 4.3. Coupled NARSES-AENEID methodology for calculating the fine scale national spatial distribution of NH_3 emissions, allowing for spatial variation in ammonia emission potentials and abatement by implementing outputs from the NARSES system into the AENEID model.

The main differences in the NH_3 emission calculation methodology by NARSES, AENEID and the coupled NARSES-AENEID models are summarised in Table 4.3. One of the major differences between the AENEID model (Figure 3.3) and the coupled NARSES-AENEID model (Figure 4.3) is the source for calculating the emission potentials and apportioning percentages. In the AENEID model, the emission potentials are based on the Unit approach (Section 1.5.1) and were derived from IAEUK (Misselbrook *et al.,* 2004). The NARSES system is based on the Mass Flow approach, and these emission potentials are used as input to the coupled

NARSES-AENEID model. Housing, storage, spreading and grazing emissions for each of the 10 x 10 km NARSES squares were extracted from the NARSES N-flow module to calculate representative apportioning percentages and emission potentials for each NARSES square. For instance, the emission potential for sheep in a particular 10 x 10 km grid cell was derived by dividing the sheep emission of that square by the number of sheep reported in that grid cell. The resulting emission potential is only representative for that particular NARSES square, as regional differences in e.g. abatement, soil conditions and climate have been incorporated in the emission calculation within the NARSES N-flow module. The regional variation in emission potentials in the coupled NARSES-AENEID model is therefore represented at a resolution of 10 x 10 km. The AENEID model was modified to allow for this and uses 10 x 10 km grid data as input, instead of the parameter file.

Table 4.3. Major differences between the NARSES, AENEID and the coupled NARSES-AENEID models.

	Resolution	Agricultural Census input data	Livestock groups	Source of emission potentials	Approach
NARSES	10 km	Level-I	46 categories	Emission potentials based on the Mass-Flow Approach, and aspects affecting local emission source strength	Mass-Flow Approach
AENEID	5 km	Level-II	46 categories	IAEUK NH_3 emission inventory	Unit Approach
Coupled NARSES-AENEID	5 km	Level-I (input to the NARSES system) Level-II (input to the AENEID model)	10 Sectors (to link the two models) 46 categories (applied in the final emission calculation)	NARSES N-flow module	A combination of the Mass-Flow Approach and the Unit Approach

Both the NARSES N-flow module and the AENEID model are based on the same 46 livestock and crop categories. The coupled NARSES-AENEID model however uses 10 aggregated sectors (e.g. dairy cows, other cattle) derived from the 46 categories, to represent a common denominator to link the two models (Table 4.4), as the NARSES outputs are not provided at category level. This 'sector approach' is used to transfer information on variable emission source strength (emission potentials) from

Table 4.4. Aggregation of NARSES categories into NARSES sectors for calculation of apportioning percentages and emission potentials.

NARSES-AENEID sector	NARSES categories used in the sector aggregation for NARSES-AENEID
Dairy cows	Dairy cows & heifers Dairy heifers in calf, 2 years and over Dairy heifers in calf, less than 2 years
Other cattle	Beef cows & heifers Beef heifers in calf, 2yrs and over Beef heifers in calf, less than 2 years Bulls >2yrs Bulls 1-2yrs Other cattle, over 2yrs Other cattle, 1-2yrs Other cattle, under 1yr
Sheep & goats	Sheep Lambs, under 1 year old Goats
Pigs	Sows in pig & other sows Gilts in pig & barren sows for fattening Gilts > 50kg not yet in pig Boars Other pigs, 110kg and over Other pigs, 80-110kg Other pigs, 50-80kg Other pigs, 20-50kg Other pigs, under 20kg
Poultry – Layers	Layers
Other poultry	Breeding birds Broilers Pullets Turkeys Other poultry
Horses	Horses
Deer	Deer
Crops	13 crop categories
Grassland	Grassland under 5 years Permanent grassland

NARSES to AENEID, while NARSES-AENEID still redistributes on the basis of all 46 categories. The average emission potentials applied for each sector are still representative for each livestock category within the sector, as long as there are no major discrepancies between the non-disclosive agricultural data at a 10 x 10 km resolution in the NARSES N-flow module and the more detailed agricultural data in the NARSES-AENEID model. These discrepancies occur mainly as a result of Level-I data being manipulated in the processing into a non-disclosive format to ensure confidentiality (see Section 4.3.1 for details on how this is dealt with).

Ammonia emissions from categories belonging to the same sector are assumed to occur on the same landcover types, which is a prerequisite for applying sectors rather than categories in the NARSES-AENEID model.

4.3.3 Spatial distribution of ammonia emissions from livestock

Similar to the AENEID approach described in Chapter 3, the NARSES-AENEID model for livestock is calculated at a 1 x 1 km grid using the following input data (see Figure 4.3):

- *1 x 1 km aggregation zone dataset*
- *UK Agricultural Census data (Level-II)*
- *LCM2000*
- *Emission outputs from the NARSES N-flow module,* i.e. housing, storage, manure spreading onto arable land and grassland and grazing emissions for each sector in each 10 x 10 km NARSES grid cell. These emission calculations were based on the non-disclosive Level-I data and were used to derive spatially variable emission potentials and apportioning percentages in NARSES-AENEID.

The input data were applied in a modified AENEID model (coupled NARSES-AENEID) that can handle the calculation of spatially variable apportioning percentages at a 10 x 10 km grid resolution within the model. This replaces the parameter file in the original AENEID model, where the parameters had been calculated from Misselbrook *et al.* (2004). The emissions from livestock categories within each aggregation zone were subsequently re-distributed onto suitable land cover data following the methodology described in Chapter 3. Spatially variable emission potentials (as 10 x 10 km grid values) were then applied to calculate the final emission map.

4.3.4 Spatial distribution of ammonia emissions from fertilized crops and conserved grassland

The approach to calculate NH_3 emissions from fertilized crops and grassland in the coupled NARSES-AENEID approach is different from the original AENEID approach, where NH_3 emissions were calculated for each aggregation zone and then re-distributed onto suitable agricultural land within the zone. In the coupled NARSES-AENEID application, all crop and grass categories (Level-II) are re-distributed within each aggregation zone onto suitable landcover (arable land and grassland, respectively). Ideally, spatially variable emission potentials (emission per grass and crop type) would be extracted from the NARSES N-flow module for each 10 x 10 km grid cell for application in the coupled NARSES-AENEID model to the already distributed agricultural census data (crop and grassland categories). At present however, the NARSES N-flow module is not able to calculate spatially distributed NH_3 emissions from fertilized crops and grassland, but only the total emission for cropland and grassland for the whole country. This will be addressed by the NARSES project team in the future, to allow the N-flow module to calculate spatially distributed NH_3 emissions from fertilized crops and grassland. Due to the current state of the N-flow module, however, the total emission value derived from the module was scaled by area of arable and grassland in the UK.

4.4 Calculating ammonia emission maps

4.4.1 Comparing Level I & Level II data

As discussed previously, the coupled NARSES-AENEID model is based on both Level I and Level II data. In order to assess the differences between the two datasets, and the subsequent potential impact on the emission result, the AENEID approach (as described in Chapter 3) was applied to calculate NH_3 emission maps based on

a) Level-I data (non-disclosive 10-km census data) and

b) Level-II data (disclosive census data).

Maps were calculated at 1-km, 5-km and 10-km resolution for both Level I and Level II data. The calculation of emission maps at 1-km and 5-km grid resolution for Level-I data (10-km resolution) was achieved by using the 10 x 10 km NARSES grid cells as "aggregation zones". At Level-I, the agricultural census data were therefore distributed within each 10 x 10 km NARSES grid cell following the AENEID methodology. In contrast, at Level II the aggregation zones were represented by the parishes. This operation enabled a straightforward comparison of the results by using the two different data sets (Level-I and Level-II) as input to the same distribution methodology (the AENEID approach), the only difference being the aggregation zones (10 x 10 km grid cells and parishes).

4.4.2 Application of the coupled NARSES-AENEID model

The link between the two models to set up the coupled NARSES-AENEID model (referred to as the Desktop Link) is represented by the emission outputs from the NARSES system (10-km grid resolution) used as input to AENEID. However, in order to adequately compare the emission results of the coupled NARSES-AENEID model with the original AENEID model, differences in emission potentials were eliminated by applying the AENEID emission potentials (based on IAEUK) for the NARSES Level I data (referred to as the Functional Link). The Functional Link provided a way to back-calculate emissions in the NARSES system based on the same emission potentials applied in AENEID to enable a quantitative comparison of the two models due to differences in methodology rather than emission potentials.

1) *Desktop Link* – Total emissions for each animal husbandry stage (grazing and housing emissions, emissions from storage and spreading of manure onto arable and grassland, respectively) are extracted directly from the NARSES system at 10 x 10 km resolution and used as input to the AENEID model.
2) *Functional Link* – Total emissions for each animal husbandry stage are calculated for each 10 x 10 km NARSES grid cell based on NARSES Level-I data and the AENEID emission potentials (derived from IAEUK). These data are calculated independently from the NARSES system and are therefore not extracted from the N-flow module.

However, a disadvantage of using outputs from the N-flow module is that currently the NARSES system is not fully developed, so that some emissions (horses, deer, goats and fertilizers) cannot be extracted from NARSES. Another disadvantage of the current state of the NARSES N-flow application is that it is not possible to extract data for the whole of the UK. Instead, the data had to be extracted in two parts, one part representing the north of the UK, and one part representing the south of the UK. These problems are however expected to be addressed in the future to provide a more flexible emission model.

In this study, the coupled NARSES-AENEID model was run for Great Britain using the Functional Link approach and by applying outputs from the NARSES N-flow module (Desktop Link). The results were compared with the original AENEID model (Level-II data). Northern Ireland was excluded from the comparison because the agricultural census data are the same for Level-I and Level-II, i.e. Level-I data are aggregated directly from Level-II data.

4.5 Results and discussion

Emission maps based on different levels of detail of the input data (Level-I and Level-II) are shown in Figure 4.4 and Figure 4.5. Emission results applying the NARSES-AENEID Functional Link and the coupled NARSES-AENEID Desktop Link at a 5 x 5 km resolution are shown in Figure 4.6. All maps were calculated for the year 2000.

4.5.1 Comparison of Level-I & Level-II output from the AENEID model

Applying the AENEID model using Level-II data is considered to give the most detailed spatial modelling results, because the highest resolution agricultural census data available are used as input to the model. When comparing the emissions maps based on the different input data (10 km non-disclosive Level-I and disclosive Level-II), as shown in Figure 4.4 and Figure 4.5, the general pattern is similar. There are however some differences, with the AENEID (Level-I) map (Figure 4.4.a) showing a smoother pattern with fewer emission peaks compared with the AENEID (Level-II) map (Figure 4.4.b). This was expected as the Level-I data were amalgamated more widely to ensure non-disclosivity in the input data than the Level-II data. In the 5-km AENEID (Level-I) map (Figure 4.4.a), where the emissions were re-distributed at 1-km resolution within the 10 x 10 km grid cell and then aggregated into 5-km resolution, it is still possible to detect the underlying 10 x 10 km squares that the re-distribution is based on. Re-distributed data based on grid square aggregation are relatively easy to detect, whereas the underlying aggregation zones (parishes) used in the Level-II approach (Figure 4.4.b) are more difficult to detect visually.

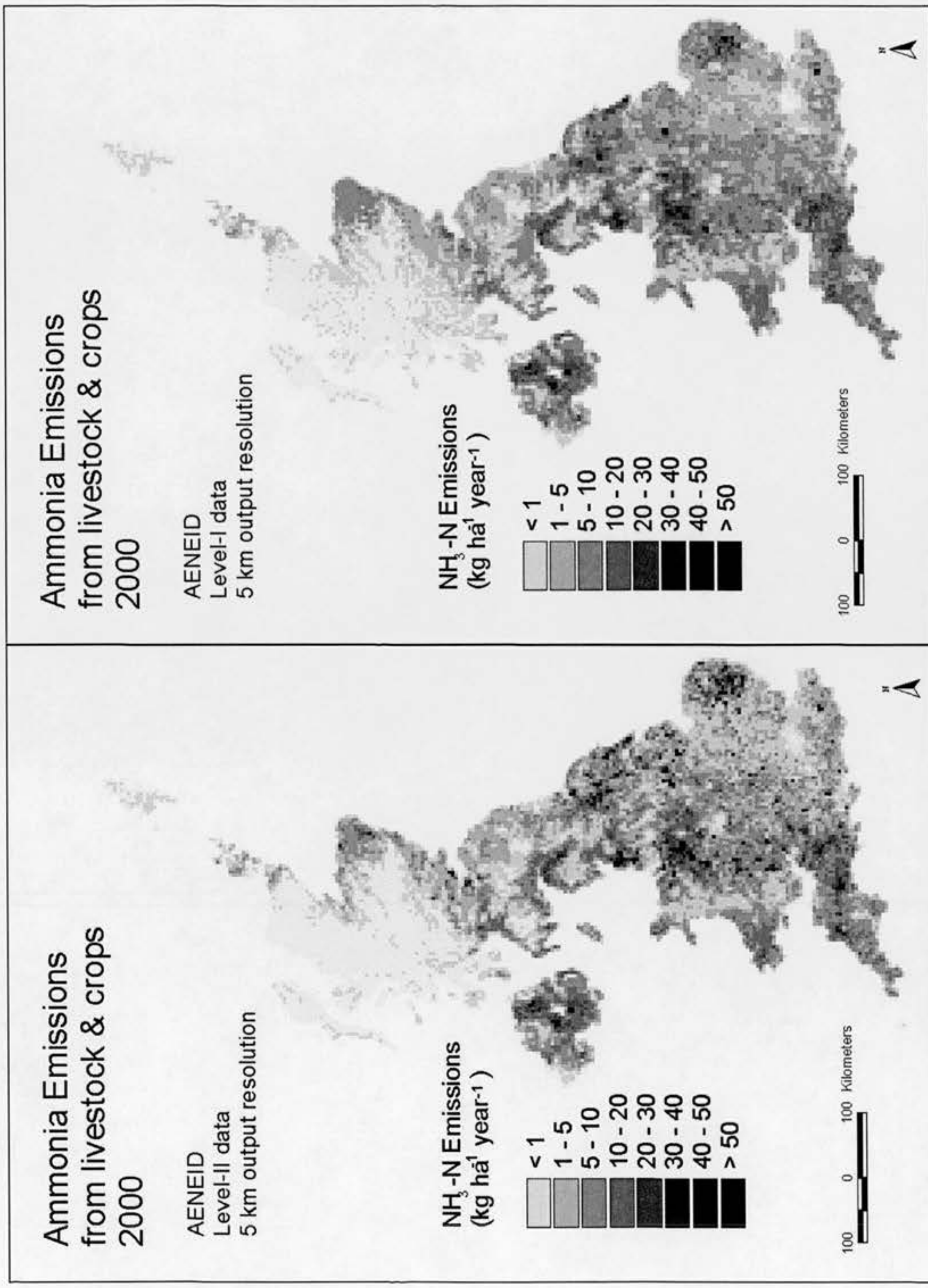

Figure 4.4. a and b: 5-km resolution ammonia emission maps, based on a) Level-I (Agricultural census input resolution of 10 x 10 km) & b) Level-II data (Agricultural census input resolution at parish level) to the AENEID model running at 1-km grid resolution and then aggregated into 5-km output resolution.

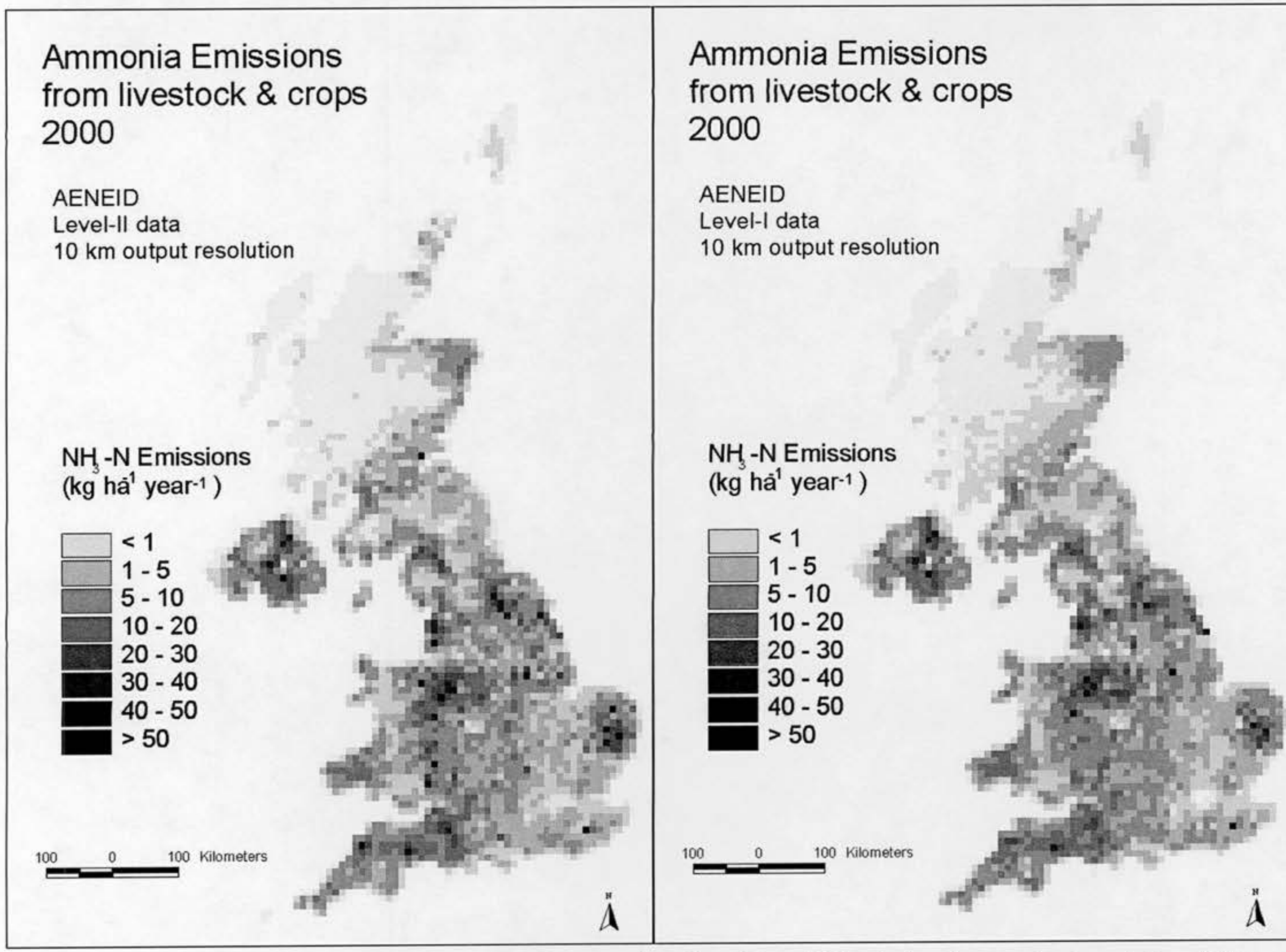

Figure 4.5. a and b: 10-km resolution ammonia emission maps, based on a) Level-I (Agricultural census input resolution of 10 x 10 km) & b) Level-II data (Agricultural census input resolution at parish level) to the AENEID model running at 1-km grid resolution and then aggregated into 10-km output resolution.

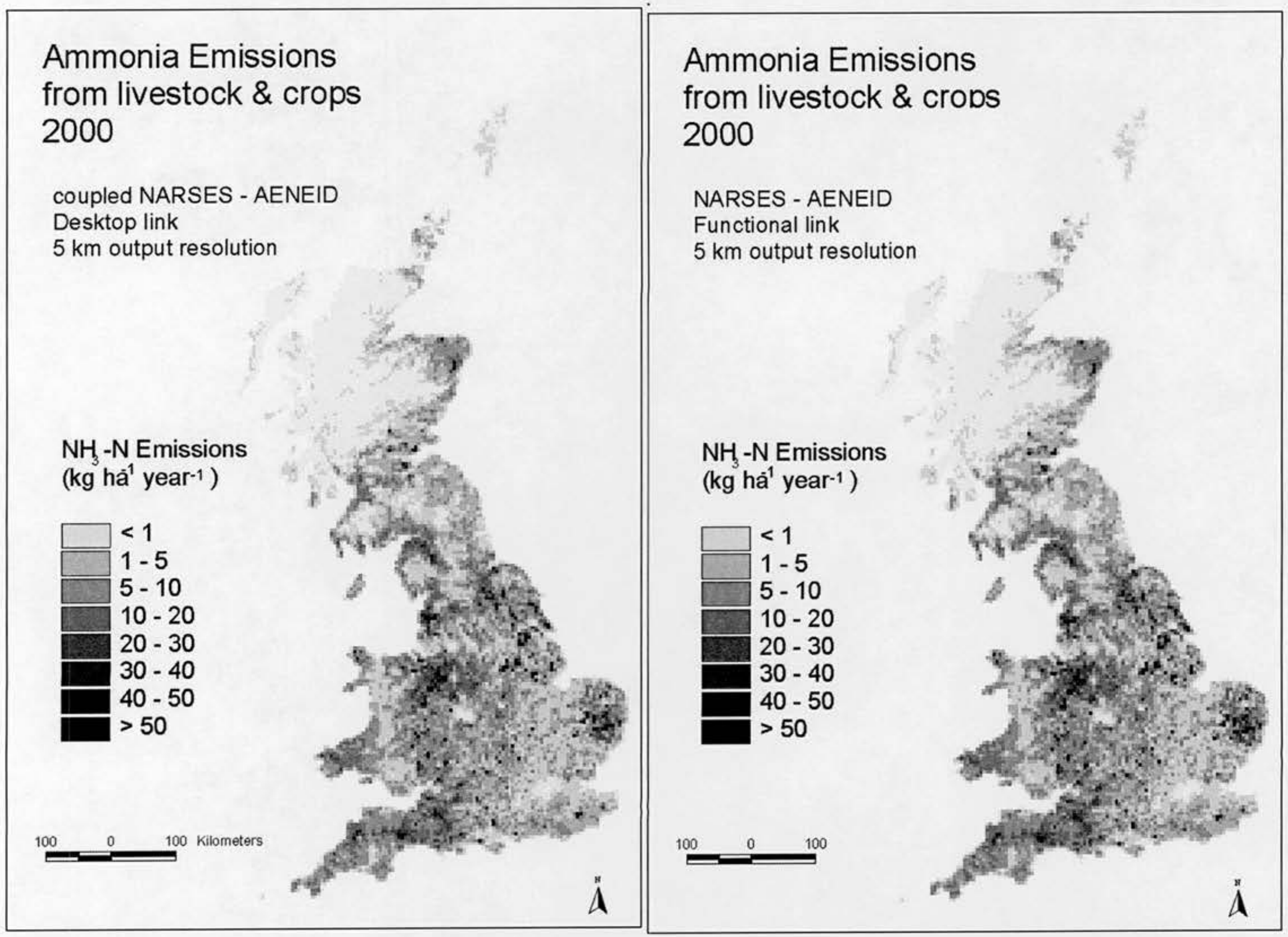

Figure 4.6. a NARSES – AENEID Functional Link at 5-km resolution and b. Coupled NARSES-AENEID Desktop Link at 5-km resolution.

Table 4.5 summarises the emission results based on the different model calculations applied in this study. As expected, Level-I results were generally smaller as confidential data deemed as identifiable may have simply been excluded by EUDL. In most cases, the overall difference was very small (< 1 %), but there were some exceptions: turkeys (99 %), other poultry (54 %), horses (79 %), goats (13 %), deer (80 %) and an unexpected difference of 556 % for one of the pig categories, i.e. one pig category was almost six times greater in the Level-I data compared with Level-II. The main reason for these large differences is that the non-disclosive Level-I data do not report any livestock numbers at all for these categories for some of the devolved regions in the UK. Level-I data do not report turkeys or 'other poultry' (i.e. geese, ducks etc) for England and Wales, horses are not reported in England and farmed deer are only reported for Scotland. The larger result for one of the pig categories in Level I than Level II was however more of a mystery. When this was investigated further, it was found that this was due to an error in the Level-I census data used in the NARSES N-flow module. It was found that one of the pig categories (n15) had been applied both as n14 and n15. This resulted in about six times more pigs being reported for n14 than should have been the case. This error has implications for the map calculated using Level-I data, with higher than expected pig emissions. This error is currently being addressed in the NARSES system, and should be corrected in due course to provide more accurate emission results in the future.

The emission results from Level-I and Level-II data in Table 4.5 are generally smaller in each sector for Level-I data, with the exception of pigs (+ 9.7 %, due to the error discussed above). The missing data for turkeys, other poultry, horses and deer also have a large impact on the emission calculation for those sectors, with smaller emissions than expected. These sectors are however relatively small in terms of the total UK ammonia emissions, and they therefore only have a small impact on the total emission result. AENEID Level-I emissions for cattle and sheep are similar to Level-II emission results (the difference is < 1 %), which indicates a good overall agreement between Level-I and Level-II data for these categories. Overall, the NH_3 emission calculation based on Level-I data estimates 3.8 % lower emissions than for the Level-II data.

Table 4.5. Emission results for Great Britain (2000) from a) AENEID using Level-II census data (5 km), b) AENEID using Level-I data (5 km), c) NARSES-AENEID Functional Link (5 km), based on AENEID emission potentials applied in NARSES, and d) coupled NARSES-AENEID (Desktop Link, 5 km), where outputs from the NARSES N-flow module were applied in AENEID. Emission differences in percentages for b) c) and d) compared with a) are also presented.

NARSES-sector	a) AENEID Level-II data ($kg\ NH_3\ year^{-1}$)	b) AENEID Level-I data ($kg\ NH_3\ year^{-1}$)	Difference between a) and b)	c) NARSES-AENEID Functional link ($kg\ NH_3\ year^{-1}$)	Difference between a) and c)	d) NARSES-AENEID Applying outputs from the NARSES N module ($kg\ NH_3\ year^{-1}$)	Difference between a) and d)
Cattle	120.14	119.89	-0.2%	120.11	0.0%	120.28	0.1 %
Sheep	16.88	16.84	-0.3%	16.89	0.1%	16.01	-5.2 %
Pigs	27.84	30.54	9.7%	28.71	3.1%	29.78	7.0 %
Poultry	42.58	34.91	-18.0%	39.23	-7.7%	35.75	-16.0%
Horses	3.53	0.75	-78.9%	3.53	0.0%	*	
Deer	0.05	0.01	-79.7%	0.05	0.0%	*	
Livestock emissions ($kg\ NH_3\ year^{-1}$)	**211.03**	**202.93**	**-3.8%**	**208.58**	**-1.2%**	**201.82**	**-4.4 %**
Crop	14.48	14.46	-0.1%	14.48	0.0%	*	
Grass	10.37	10.31	-0.5%	10.37	0.0%	*	
Fertilizer emissions ($kg\ NH_3\ year^{-1}$)	**24.85**	**24.77**	**-0.3%**	**24.85**	**0.0%**	*	
Total emission ($kg\ NH_3\ year^{-1}$)	**235.88**	**227.71**	**-3.5%**	**233.42**	**-1.0%**	**226.66****	**-3.9 %**

** Values not feasable as output not yet possible from the NARSES system*

*** Emission value if values from AENEID (Level-II data) were applied.*

While Table 4.5 provides a means to compare overall emissions, it fails to capture any spatial differences between the different emission approaches. Although the overall emission of Level-I and Level-II data is similar, the difference regarding the spatial distribution of emissions is far greater. Level-I data (Figure 4.4.a and Figure 4.5.a) show a much smoother emission pattern with few emission peaks compared with Level-II (Figure 4.4.b and Figure 4.5.b). Spatial detail is lost when the smoother Level-I data are applied and the spatial representation error in these maps is therefore greater, particularly for those livestock categories where confidentiality is important (pigs and poultry). Furthermore, Level-I data is only available at 10-km resolution (Figure 4.4.a) and the 5-km map based on Level-I data (Figure 4.4.a) was only made possible through the AENEID approach, where the NARSES categories were re-distributed at 1-km resolution within the 10-km grid cell, and thereafter aggregated into 5-km resolution. The highest resolution available for the final NH_3 emission calculation in the NARSES N-flow module is therefore at 10 x 10 km grids, while the AENEID approach enables emission calculations at a much higher resolution.

4.5.2 NARSES-AENEID model – Functional link

A good correlation between the two agricultural data sets (Level-I and Level-II) is vital in the coupled NARSES–AENEID model, where both Level-I data and Level-II data are applied. When comparing NARSES-AENEID (Functional Link) results (Figure 4.6.a) with the AENEID (Level II) output (Figure 4.4.b), the maps correspond very well, in fact, they are practically identical. This is because the NARSES-AENEID (Functional Link) approach is based on the same emission potentials (derived from IAEUK) as the AENEID approach (Level-II). The only difference is that the emission potentials and apportioning percentages applied in the Functional Link are sector based at a 10 km resolution, instead of representing the average UK value for each individual category as is the case for the AENEID approach (Level-II).

The purpose of calculating emissions using the Functional Link approach was to estimate the possible uncertainties associated with *the methodology* of the coupled NARSES-AENEID approach; i.e. a sensitivity analysis to investigate how the methodology (applying sector based emission potentials and apportioning percentages at 10 km resolution) affects the emission result was performed. In the Functional Link approach spatially varying emission potentials were therefore not applied in the calculation. Comparing the two approaches of AENEID and NARSES-AENEID (Figure 4.4.b and Figure 4.6.a) therefore serves to demonstrate that there is sufficient correlation between the Level-I and Level-II datasets to allow the operation of the coupled NARSES-AENEID approach.

Some of the 10 x 10 km squares used as input to the NARSES-AENEID model did not contain any data for a particular category at Level-I, e.g. deer in England and Wales. For these squares, a default average emission potential was applied. This default value may be based on either national numbers, or on neighbouring 10 x 10 km grid squares. The latter would probably be more realistic, as it would reflect local conditions better. However, further work is necessary to improve and streamline the handling of such differences. Applying an average value contributed to improving the emission result in the NARSES-AENEID approach (Column c in Table 4.5) compared with AENEID (Column a in Table 4.5), although the correlation between Level-I and Level-II (defined as the percentage difference between (Columns a) and (b) in Table 4.5) data was unsatisfactory.

4.5.3 Coupled NARSES-AENEID model – Desktop Link

The coupled NARSES-AENEID approach applying output from the NARSES-N-flow module could only be carried out for some of the livestock sectors, because currently it is not possible to extract emissions from horses, goats and deer from the NARSES N-flow module. Fertilizer emissions could not be calculated either, because the fertilizer model in the NARSES N-flow module has not yet been fully developed. Due to these limitations, fertilizer emissions presented here are based entirely on AENEID (Level-II).

When comparing the emission results of the coupled NARSES-AENEID (Desktop Link) (Figure 4.6.b) with the AENEID model based on Level-II data (Figure 4.4.b), the maps look very similar. However, in a quantitative comparison of the two approaches (in Table 4.5), it becomes clear that the two approaches arrive at different emission results. Cattle show a very good correlation (0.1 %). Emissions from pigs, on the other hand, differ by 7 % between the two approaches, however this is mainly due to the incorrect pig category in the NARSES N-flow module. Emissions from both sheep and poultry differ significantly from the AENEID (Level-II) approach (by 5.2 % and 16 % respectively). The main explanation for these large differences is that the emission potentials in the N-flow module have been converted from the Unit approach of Misselbrook *et al.* (2004) into the proportion of available TAN (the Mass-Flow approach) of Webb *et al.* (2004).

As the NARSES N-flow module is based on the Mass-Flow approach and allows the incorporation of spatially varying constraints and factors in the 10 x 10 km grid cells, it also allows for potential abatement scenarios to be calculated. These regionally varying emission potentials and abatement scenarios have not yet been fully implemented in the model, and the NARSES emission result can therefore currently be seen as the effect of converting the Unit-approach emission potentials of Misselbrook *et al.* (2004) into the Mass-Flow approach of Webb *et al.* (2004). This present study however does serve to demonstrate that, once regional emission potentials have been incorporated fully in the NARSES N-flow module, the AENEID model will be able to apply these spatially variable data when calculating fine-resolution (1-km) NH_3 emission maps.

4.6 Conclusions

Ammonia emission estimates based on different levels of detail of input data (Level-I and Level-II) were calculated for the whole of UK. The emission estimates based on the less-detailed non-disclosive (Level-I) census data are associated with larger uncertainties than estimates based on the more detailed confidential (Level-II) census data. This is due to the processing of Level-I data into a non-disclosive format to ensure confidentiality. The non-disclosive format of Level-I data have, however, the advantage that they can be applied without any confidentiality constraints, which makes it possible to apply them in the NARSES N-flow module which can be used by Defra staff. Calculating ammonia emissions in the NARSES system makes it possible in principle to integrate abatement scenarios and regional variations in emission source strength. These spatially varying emission potentials can then be extracted from the NARSES N-flow module and applied to the spatially distributed (detailed) Level-II data in the coupled NARSES-AENEID approach.

This study has demonstrated the feasibility of operating a coupled NARSES-AENEID model, i.e. using output from the NARSES N-flow module as input to the AENEID model. The coupled NARSES-AENEID emission map shows a good estimate of NH_3 emissions where the two data sets (Level-I & Level-II) showed a good agreement. For livestock categories that do not correspond well, the agreement could be improved by imputing values for missing (Level-I) data, for example by extracting emission data from neighbouring 10 x 10 km NARSES grid cells.

The advantage of the NARSES N-flow module is that in due course it will be possible to calculate NH_3 emission maps based on variable emission potentials. Conversely a disadvantage is the low output resolution of 10 x 10 km and dependence on Level I (non-confidential) data. Due to the confidentiality constraints of the Level I data, 10 x 10 km resolution is the best possible resolution available for the purposes of NARSES. The AENEID approach on the other hand allows for much higher resolution output and use of confidential Level II data, but is not based on regionally varying emission potentials. By linking the two models and applying the coupled NARSES-AENEID model, the strengths of both models can be integrated to generate a superior model that takes advantage of both the high resolution modelling and the spatially variable emission potentials.

At present, the link between the two systems is limited, due to constraints in the NARSES software. In order for the coupled model to be more efficient, it is necessary to be able to extract data for the whole of the UK in one single operation and in a more streamlined fashion. Subsequent work by the NARSES project team should focus in particular on improving these linkages, and further development of the NARSES N-flow module to incorporate a larger degree of spatial variability in emission potentials. Furthermore, the performance of the fine scale modelling (NARSES-AENEID) should be tested against measurement data.

5 Modelling seasonal dynamics in spatially resolved ammonia emissions for the UK

5.1 Introduction

Most ammonia emission inventories have been calculated on an annual basis and do not take into account the seasonal variability of emissions that occur as a consequence of climate and agricultural practices that change throughout the year. When used as input to atmospheric transport models to simulate concentration fields, these models therefore fail to capture seasonal variations in ammonia concentration and dry and wet deposition.

However, recently, NH_3 emission models with a temporal element have started to emerge (Hutchings *et al.*, 2001; Battye *et al.*, 2003; NARSES, 2004; Pinder *et al.*, 2004; Skjøth *et al.*, 2004; Gyldenkærne *et al.*, 2005), In the present study, an approach to calculate monthly NH_3 emissions has been developed at a 5-km grid resolution (Hellsten *et al.*, 2006; Sutton *et al.*, 2006). The approach is based on the AENEID model and calculates "snap-shots" of the spatial distribution of emissions for each month by incorporating temporal activity data. The model incorporates changes both in the spatial location of sources and the magnitude of NH_3 emissions due to the seasonality of agricultural practice.

5.2 Seasonal variations in ammonia emissions

Many studies where NH_3 concentrations have been measured have shown large daily, monthly and annual variations in concentration (Yamamoto *et al.*, 1995; Horvath and Sutton, 1998; Milford *et al.*, 2001; Huber and Kreutzer, 2002; Sutton *et al.*, 2003). These temporal variations may be influenced by both background concentrations and local sources. Generally, measurements of NH_3 at a location over several years show similar general patterns, but with considerable variability between years. Many studies have shown that the temporal variation in NH_3 concentrations is to a large degree affected by temperature, with higher NH_3 concentrations in summer, and lower concentrations in winter (Yamamoto *et al.*, 1995; Horvath and

Sutton, 1998). Huber and Kreutzer (2002) showed that temperature is important at a diurnal resolution, with higher concentrations in the late morning and lower concentrations during the night. Burkhardt *et al.* (1998) on the other hand did not find a correlation with air temperature when measuring during 2 years at a rural site close to Edinburgh. They suggested that the lack of correlation with temperature was because the measurements were to a large degree affected by local sources, and that the temperature dependence would have been clearer if the site had been dominated by background concentrations.

Measurements have also shown a clear link between ammonia emissions and farming practices. Horvath and Sutton (1998) recorded spring and autumn peaks from long-term years of measurements in Hungary and suggested that these peaks were explained by increased agricultural activity during these seasons. Huber and Kreutzer (2002) also recorded higher NH_3 concentrations during spring and autumn in Bavaria, Germany, which they attributed to agricultural activities, such as manure application and fertilization. These emissions are very season-dependent, since more manure and fertilizers are applied in autumn and spring than in winter and summer (BSFP, 2001; Scott *et al.*, 2002).

A seasonal pattern of aerosol ammonium (NH_4^+) concentrations has been more difficult to detect from measurements than seasonal NH_3 patterns. Horvath and Sutton (1998) however recorded a seasonal pattern, with larger concentrations in winter and smaller concentrations in summer, but this trend was not as clear as for NH_3 concentrations. Other studies, e.g. Yamamoto *et al.* (1995), have recorded rather constant NH_4^+ concentrations during the year, without significant seasonal variations. NH_3 occurs in the atmosphere as a result of direct emissions, but the occurrence of NH_4^+ aerosols is very dependent on meteorology and reactions with acids.

The National Ammonia Monitoring Network (NAMN) in the UK (Section 1.5.4) assess temporal trends of concentrations, both long-term, intra-annual trends and inter-annual trends (Figure 5.1). The monitoring sites in the NAMN (Figure 1.17) are located away from point sources in order to receive a representative value for the area. The measured NH_3 concentrations therefore show a relatively smooth pattern. However, the measurements serve to demonstrate that seasonal variations in NH_3

occur during a year, and that these variations are influenced by both meteorological conditions (such as temperature) and farming practice.

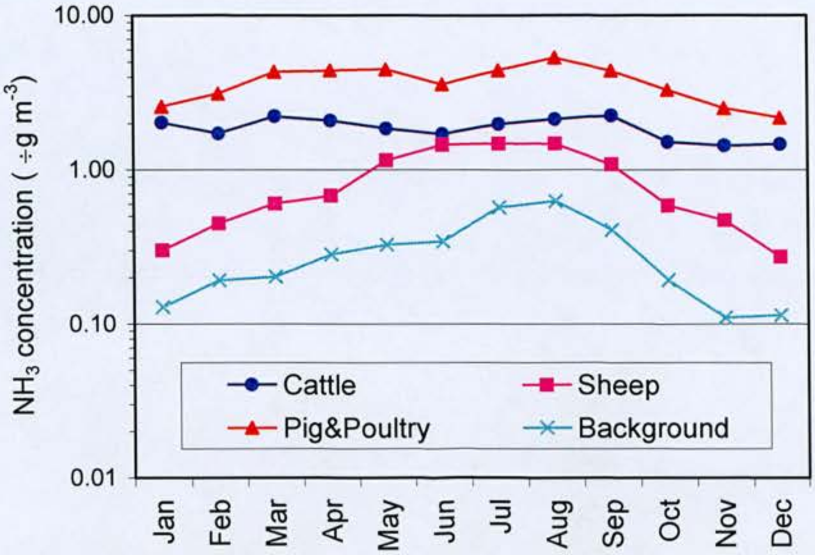

Figure 5.1. NH_3 concentrations from the mean data for 83 selected sites in the NAMN across the UK (1996-2002), classified as dominated by cattle, sheep, pig & poultry or background emissions. Source: Tang and Sutton (2004).

5.3 Approaches to include temporal aspects in GIS

Temporal aspects in Geographical Information Systems are a complicated issue, because they introduce another dimension in addition to the spatial context. The temporal dimension also tends to be more difficult to define than the spatial dimension.

The most simplistic and traditional way to incorporate a temporal dimension in GIS is to provide *snapshot maps,* where each image represents the current state at a given point in time. This is a static representation of reality which is simple to derive. The main disadvantage of this approach is that it does not maintain any temporal structures or relationships (Langran, 1992). As a consequence, interesting phenomena may be "hidden" in-between the snapshots. Another disadvantage is that the snap-shot approach stores redundant information, because all the unchanged data are duplicated at each snapshot. Major advantages of the snap-shot model are, however, that it retains the simplicity of the raster model and that it is easy to retrieve the entire state at each snapshot (Peuquet, 2001).

Another way to incorporate time in spatial analysis is by applying a *base map with amendments.* In this approach, a base map represents the initial state, and changes are recorded when they occur (Langran, 1992). This is a more complex model than the snap-shot approach, but an advantage is that the volume of data is reduced because only the changes from the base state are stored, rather than a complete new snapshot.

The *space composite* approach is another approach to incorporate a temporal dimension in a GIS. In this approach, the geographical area affected by change becomes a distinct object, i.e. the image decomposes over time into smaller fragments (Langran, 1992). As is the case with the base map with amendments, this approach also retains the temporal structures and relationships, while minimising the storage of redundant data. A disadvantage, however, is the increasing amount of small objects to handle.

Another approach is the *space-time cube*, where the spatio-temporal objects are represented by a cube with three dimensions, one dimension representing time and the other two dimensions representing space (Langran, 1992). This approach demands significant data storage and processing, and is increasingly complicated if a fourth dimension, e.g. topology, is introduced. Peuquet (2001) suggests that the snapshot-approach can actually be viewed as a 3-D space-time cube, where each raster cell contains the value of the attribute data at the time of the snapshot.

More recently, *object-oriented, spatial-temporal data models* have been developed to incorporate time in spatial analysis. For instance, Li and Cai (2002) developed a model that was able to link space and time, because all entities were modelled as objects that contain a class-id, attributes (theme, space and time), and behavioural information (spatial and temporal relationship with other objects and spatial change of the object). This approach is complex, but has the advantage of being able to query time as well as space, because space and time are linked together and changes can be stored both in spatial and temporal dimensions.

The choice of approach for implementing time into a GIS model depends on the task at hand, the input data and the type of temporal information available. Spatio-temporal data are often very limited because they are either not available at frequent enough intervals, or non-existent. For the purpose of calculating seasonal NH_3 emissions, temporal activity data for agricultural source activities (livestock grazing and/or housing, manure storage and manure application) were dis-aggregated into a monthly temporal resolution (see Table 5.1). This work represented an extension to the IAEUK (Misselbrook *et al.*, 2000; 2004). The activity data incorporate temporal variations in farming practice during a year, e.g. number of grazing/housing days per month, percentage slurry and FYM spread to grass and/or arable land per month and number of manure storing days per month. For instance, lowland sheep are assumed to be partly housed during the lambing season (February to April), and the subsequent storage emissions are assumed to occur from May to July, with spreading of the manure assumed to occur from July to October. The availability of monthly activity data for this study justifies the implementation of the snap shot approach to calculate seasonal variations in NH_3 emissions.

Table 5.1. Example of temporal activity data for sheep. From T. Misselbrook, IGER, Pers. Comm., 2003.

Sheep		**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Grazing (d)	Upland sheep	31	28	31	30	31	30	31	31	30	31	30	31
	Upland lambs			31	30	31	30	31	30				
	Lowland sheep	31	18	21	20	31	30	31	31	30	31	30	31
	Lowland lambs	10	15	25	30	31	30	24	18				
Landspreading (%)	FYM							25	25	25	25		
Housing (d)	Sheep		10	10	10								
Storage (d)	FYM					30	30	30					

5.4 Incorporating seasonal variability in the AENEID model

The central challenge is how to incorporate temporal change effects into the spatial data. Changes in NH_3 emissions with time vary in their spatial location and their magnitude. For instance, cattle may graze some distance from the cattle houses in summer, but be in or near the animal houses for the rest of the year. In order to incorporate these changes in the AENEID model, it is necessary to consider three levels of temporal change:

1. *Changes with time in the spatial data* (landcover and zonal boundaries)
2. *Changes with time in the attribute data* (agricultural census data and emission potentials)
3. *Changes with time in the modelling parameters* (apportioning percentages)

These changes occur on a monthly or annual basis depending on the temporal scale applied in the model. In this study, annual NH_3 emissions for 1990, 1996 and 2000 were calculated (Chapter 8), to demonstrate *inter-annual* variability of emissions, and monthly emission maps were calculated in this chapter for 2000 to demonstrate the *seasonal variations* of emissions during one year. The temporal resolution for many of the data sets applied in this study is annual, and where possible, these data sets were disaggregated to a monthly resolution for application in the temporal AENEID model.

5.4.1 Variation in the spatial data

The spatial characteristics of landcover and zonal aggregation boundaries may change with time. Changes of landcover during the 10 year period (1990 – 2000) are not taken into account due to a lack of data to represent the change. As discussed in Chapter 3, the quality and approach taken for the landcover map from 1990 were deemed as too different from the landcover map from 2000, LCM2000 (Fuller *et al.*, 2003), and therefore LCM2000 was applied for all years.

The aggregation zones for the agricultural census data (e.g. the parish boundaries) may also change during the 10 year period. This change was indirectly incorporated in the model by applying what Blake *et al.* (2000) refer to as the "freeze history approach", i.e. the zoning system is frozen at a particular point in time. In this study, the same zonal dataset from 1996 was applied for all years, but changes in the agricultural census for the years 2000 and 1990 were adjusted to the "frozen" 1996 zoning system. This was achieved by aggregating some of the agricultural census statistics to the zonal boundaries for the 1996 dataset. Blake *et al.* (2000) point out that the main disadvantage of the "freeze history approach" is that it is inflexible, because the frozen zones become less accurate with time. The change in zonal boundaries (parishes) over a 10-year period is however small, and therefore not considered a major problem here.

5.4.2 Variation in the attribute data

Changes over time in the attribute data in the AENEID model mainly refer to changing numbers of livestock and area of agricultural land in the agricultural census data, but also to potential changes in the emission potentials. The agricultural census data are only available as annual averages (snap-shots in June), and these data were therefore assumed to be representative for the whole year, even when modelling monthly emissions. Variations in livestock numbers on a monthly basis are, however, deemed as being small for most census categories, with the exception of turkeys and lambs, which have a distinct seasonal cycle.

Changes in emission potentials from month to month are likely to have a large impact on emissions due to seasonal changes in climatic conditions and agricultural activities during the year. Changes due to agricultural practice have been accounted for by distinguishing the annual emission potentials into monthly emission potentials by incorporating temporal activity data, as shown in Table 5.1. However, these data do not account for climatic differences.

5.4.3 Variation in the modelling parameters

The rules on how to apportion the agricultural statistics to the different landcover categories ("apportioning percentages", see Section 3.4) depend on the emission source strength for each animal husbandry stage (housing, manure storage, spreading of manure, grazing livestock). As the proportion of emissions from each of these stages changes during the year, so do the apportioning percentages, and hence the distribution of the emission within the parish. For instance, the proportion of emissions allocated to those landcover types where grazing occurs (i.e. different quality types of grassland) is higher during the grazing season (summer), than when cattle are housed. Monthly apportioning percentages were therefore calculated based on the emission for each animal husbandry stage, derived from the monthly emission calculations of IAEUK.

5.4.4 Incorporating temporal activity data in the AENEID model

The development of monthly distributions of NH_3 emissions demands a more detailed level of data than at an annual level. Some NH_3 sources, such as the spreading of manure, may have a complex temporal emission pattern, which is difficult to differentiate in an inventory when data are sparse, while other sources are easier to model. For instance, housing emissions from poultry are practically constant during any annual cycle (if temperature is not taken into account).

Misselbrook (2003) provided temporal activity data on a monthly basis, '*Temporal disaggregation of inventory activity data*' (Table 5.1). These activity data incorporate temporal variations in farming practice during a year, e.g. number of grazing days per month, number of housing days per month, percentage slurry and FYM spread to grass and/or arable land per month and number of manure storing days per month.

5.5 Calculation of monthly ammonia emissions maps

The methodology used in the monthly version of AENEID is the same as in the original AENEID model, with the only difference being the temporal dimension. The temporal model applies apportioning percentages and emission potentials representative for each month rather than the annual values. The NH_3 emissions of these monthly maps differ both spatially and in magnitude, as a consequence of applying monthly varying apportioning percentages and emission potentials. For instance, cattle emissions during the winter months are concentrated on those landcover types where the cattle houses are most likely to be situated, while cattle emissions during the summer are allocated to land cover types where grazing is assumed to occur.

The monthly breakdown of NH_3 emissions from fertilizers are based on statistics of fertilizer application per month (BSFP, 2001). The monthly fertilizer emission was expressed as a percentage of the annual emission, and these proportions were then spatially distributed onto arable and improved grassland, respectively.

5.6 Non-agricultural sources

The monthly AENEID model is limited to calculate temporal emissions for agricultural sources, but in order to validate the model with measurements (see Section 5.7.6), the emission contribution from non-agricultural sources has to be included in the emission maps.

The seasonal pattern for most non-agricultural sources is, in contrast to most agricultural sources, expected to be relatively evenly distributed throughout the year (humans, pets, sewage works, transport, landfill sites, waste incineration, household products etc.). Exceptions to this are emissions from seabird colonies and non-agricultural fertilizers. These sources show a distinct seasonal emission pattern. For instance, seabird emissions should only be calculated during and for a while after the breeding season (Wilson *et al.*, 2004). While these emissions are small in UK terms, they are significant at a local scale, especially in remote areas with little agriculture where seabirds may be the main source of NH_3 emissions (Wilson *et al.*, 2004).

Overall, non-agricultural emissions are much smaller (44.7 kt NH_3-N yr^{-1}) (Dragosits and Sutton, 2003) than the agricultural sources (206.9 kt NH_3-N yr^{-1}) (Misselbrook et al., 2003), and therefore the few sources that are temporally variable, were considered not to contribute to major seasonal variations in emissions over most of the UK. Non-agricultural sources were therefore spatially dis-aggregated evenly over the year.

5.7 Results and discussion

The ammonia emissions for the different months (2000) are shown in Figure 5.2 and Table 5.2. Figure 5.2 shows a clear emission peak in March, mainly as a result of mineral fertilization application in spring and livestock being housed. Emissions then decrease from April to May due to the start of the grazing season and a decrease in fertilizer application. The emissions are estimated to be small during the summer when the livestock are grazing outdoors (particularly due to cattle grazing, as sheep tend to be outside all year round, while most pigs and poultry are housed all year round). The estimated emissions increase again in August and continue to rise towards a small peak in October, as a result of spreading of manure and livestock going back indoors. Emissions then decrease again due to less manure being applied to the fields in wintertime. Cattle are a major source of ammonia emissions (> 50 % of the agricultural NH_3 emission), and the housing/grazing pattern for cattle therefore significantly influences the overall emission pattern.

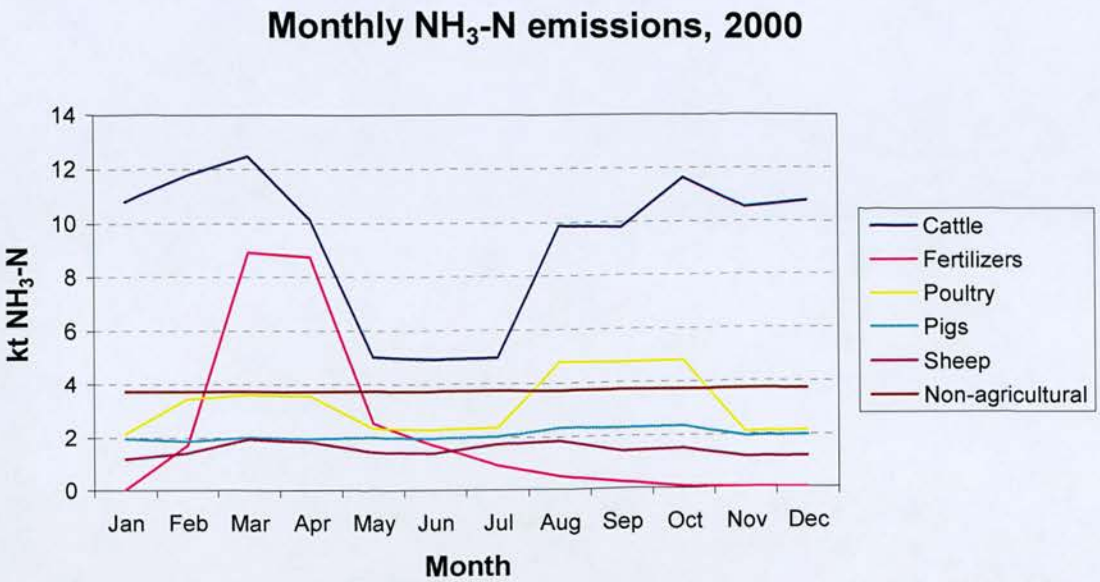

Figure 5.2. Monthly modelled NH_3-N emissions (kt) in the UK for the year 2000.

Table 5.2. Total ammonia emissions for the UK (2000), calculated from the monthly AENEID model. Non-agricultural emissions are based on 2002 (Dragosits and Sutton, 2003; Dragosits *et al.*, 2004).

NH_3-N (t)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Agricult. emissions	16,115	20,181	28,947	26,199	13,291	12,220	11,912	19,339	18,464	20,338	15,695	16,050	218,751
Fertilizers	10	1,692	8,932	8,748	2,520	1,679	914	497	240	69	14	5	25,319
Livestock	16,105	18,489	20,016	17,451	10,771	10,541	10,998	18,842	18,223	20,269	15,681	16,045	193,432
Cattle	10,816	11,776	12,481	10,127	5,006	4,918	4,954	9,876	9,793	11,641	10,529	10,758	112,674
Sheep	1,184	1,404	1,931	1,818	1,427	1,384	1,706	1,836	1,398	1,497	1,146	1,182	17,913
Pigs	1,962	1,855	2,003	1,954	1,998	1,949	1,998	2,327	2,278	2,327	1,913	1,962	24,525
Poultry	2,143	3,454	3,601	3,552	2,340	2,291	2,340	4,803	4,754	4,803	2,094	2,143	38,321
Non-agri. emissions	3,725	3,725	3,725	3,725	3,725	3,725	3,725	3,725	3,725	3,725	3,725	3,725	44,700
Total emission	19,840	23,906	32,672	29,924	17,016	15,945	15,637	23,064	22,189	24,063	19,420	19,775	263,451

5.7.1 Cattle

Cattle emissions show a strong seasonal pattern in the modelled emission, as shown in Figure 5.2. Modelled NH_3 emissions from cattle vary significantly throughout the year, with summer emissions being about half of the emissions during the spring.

This emission pattern is mainly associated with the housing and grazing season, because emissions from cattle are significantly lower when cattle are grazing than when they are housed (Webb *et al.*, 2005). Misselbrook (2003) accounted for this temporal variation through the number of grazing and housing days per month. Land spreading of cattle manure is assumed to be greater during spring and autumn in the temporal activity data by Misselbrook (2003), which is why the cattle graph shows peaks of emissions at these times.

When Pinder *et al.* (2004) calculated temporally and spatially resolved ammonia emissions for dairy cows in the United States, summer emissions from dairy cattle were estimated to be higher than during winter, in contrast to the results in the present study. The higher summer emission resulted from the temperature dependance of emissions, as Pinder *et al.* (2004) incorporated climate conditions (temperature, wind speed and precipitation) in their monthly emission factors in addition to the seasonal variation in farming practices. The results of this study suggest that the climate conditions are important to incorporate also in the seasonal emission caluclation for the UK. However, the temperature dependence is not expected to be as significant in the UK due to different farming practice and climatic conditions compared with the US. For instance, dairy farms in southern and western states in the US are associated with more intensive practices where dairy cattle are confined all year around, i.e. the cattle do not graze in the summer (Pinder *et al.*, 2004). These intensive practices, together with warmer temperatures result in higher emissions in summer compared with the rest of the year.

It can be difficult to generalise the temporal variability of NH_3 emissions from cattle, because variables affecting the temporal pattern may depend on type of breed and agricultural practice. Some breeds, such as Highland cattle in Scotland, may graze all year despite harsh climatic conditions in wintertime. These type of issues are not taken into account with the current approach.

5.7.2 Pigs and poultry

Pigs do not show a significant seasonal emission pattern in the modelled emission result as shown in Figure 5.2. This is because the temporal activity data

(Misselbrook, 2003) is fairly constant throughout the year, even when it comes to pig manure application. The small emission variations between months are mainly a consequence of the variations in number of days per month.

Poultry on the other hand show a much stronger seasonal emission pattern than pigs. Misselbrook (2003) assumes that both these categories have fairly stable housing, storage and outdoor emissions all year around in the temporal activity data. For poultry however, the seasonal pattern of manure application is much stronger than for pigs, with a larger proportion of the manure assumed to be applied in spring and autumn.

5.7.3 Sheep

The seasonal NH_3 pattern from sheep is not as significant as for cattle, because sheep graze all year around. The modelled NH_3 emissions from sheep are fairly constant throughout the year. Some variations do occur, mainly as a consequence of the assumption that lowland sheep are being housed in the spring during the lambing period, hence giving rise to subsequent storage and spreading emissions.

5.7.4 Fertilizers

Fertilizer emissions show a strong seasonal emission pattern with a significant emission peak in springtime. This is a result of a combination of factors. Firstly, the temporal activity data for fertilizers suggest that the amount of fertilizer applied each month is highest during spring, especially in March and April. Secondly, the proportion of urea applied is the highest in March and April, while urea is assumed not to be applied at all during August to January (Misselbrook, 2003). Urea is mainly applied to winter cereal and oilseed rape (BSFP, 2001), which explains the seasonal application pattern of urea. As the ammonia volatilisation rate is much higher for urea than for other fertilizers (see Table 2.11), the results show a much higher overall emission potential from fertilizers during the spring and early summer compared with the rest of the year.

5.7.5 Monthly emission maps

Monthly emission maps for Northern Ireland are shown in Figure 5.3. UK maps of ammonia emissions for the winter (January), spring (April), summer (July) and autumn (October) for the year 2000 are shown in Figure 5.4 and Figure 5.5. The emissions are larger in spring, particularly in the pig, poultry and cattle dominated areas. When modelling temporal NH_3 emissions, it is important to differentiate between temporal differences that are spatial (i.e. the location of emissions varying with time) compared with differences in the magnitude of emissions (variations in emission source strength with time). For instance, cattle emissions are more localised in winter time, because the emissions are restricted to those landcover types where the cattle houses are assumed to be located (improved pasture). Cattle emissions are also greater overall in winter time, because the emission potential from cattle is greater when cattle are housed than when they are out on the fields grazing.

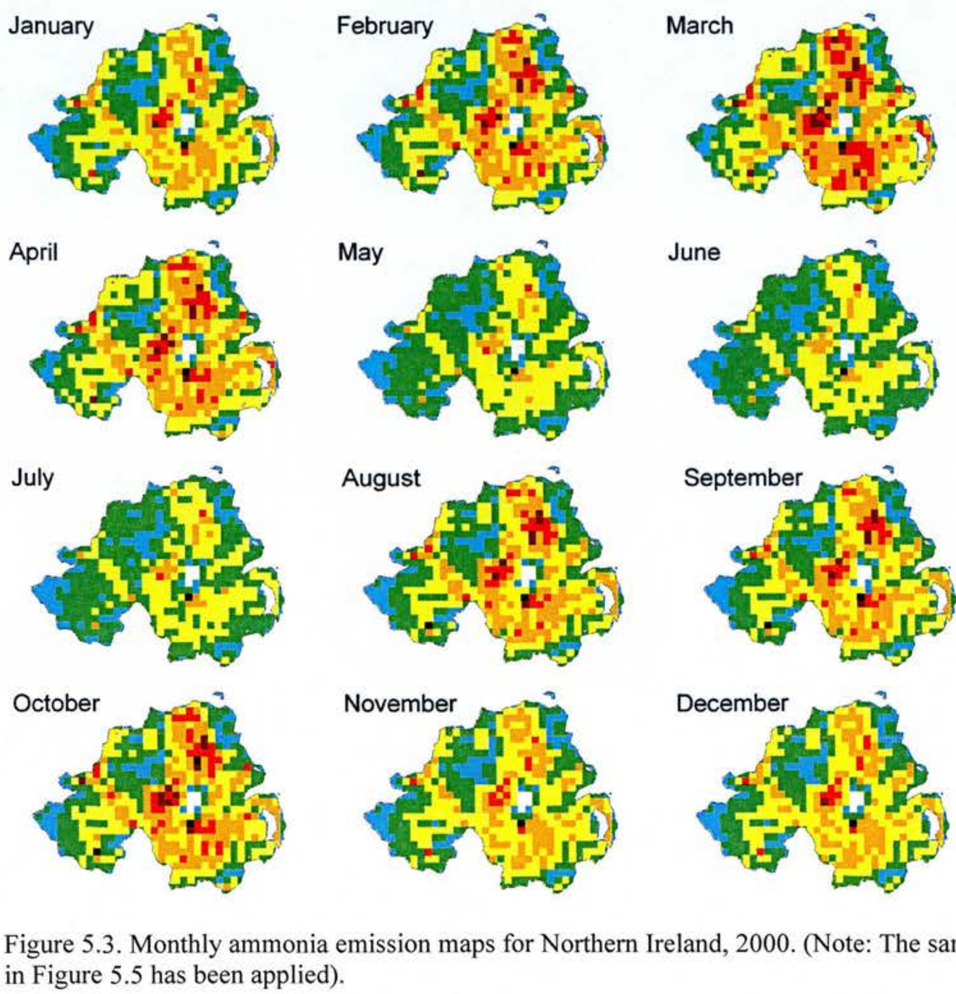

Figure 5.3. Monthly ammonia emission maps for Northern Ireland, 2000. (Note: The same legend as in Figure 5.5 has been applied).

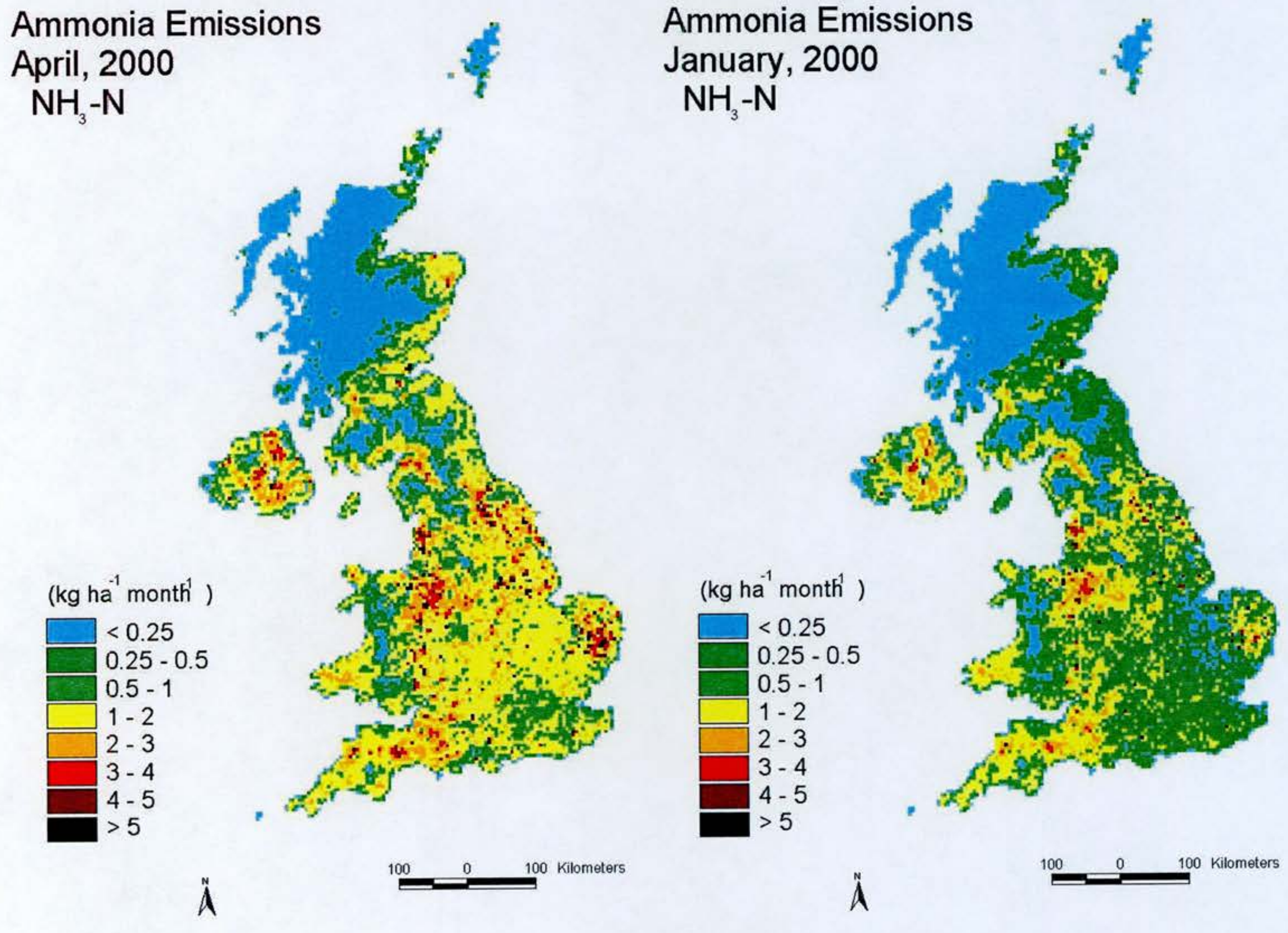

Figure 5.4. Modelled NH_3 emission maps for a winter month (January) and springtime (April), 2000.

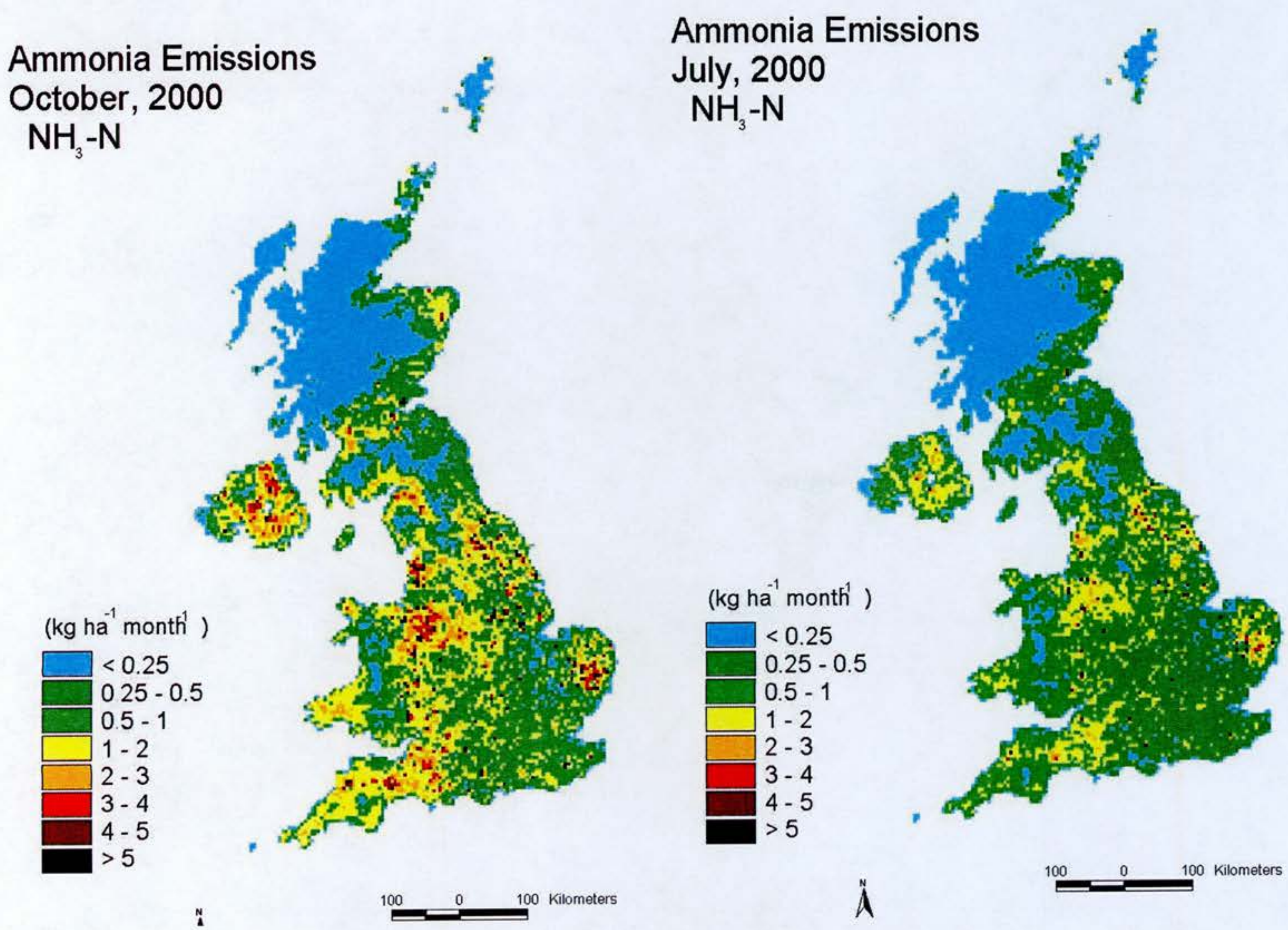

Figure 5.5 Modelled NH_3 emission maps for a summer month (July) and autumn (October), 2000.

Spatial differences between winter and summer emissions were assessed in two ways. Firstly, Figure 5.6.a shows the absolute difference in cattle emissions between January and July for 2000. Secondly, Figure 5.6.b shows the relative (%) difference between January and July, with the normalization accounting for the overall difference in UK cattle emissions between January and July. Thus, Figure 5.6.a shows the spatial variation in the overall magnitude of change, while Figure 5.6.b shows the difference in the spatial allocation of the emissions. While cattle emissions are larger in absolute terms in January than in July, (Figure 5.6.a), Figure 5.6.b shows that a higher proportion of the emissions occur in hill areas in summer, and consequently, emissions in neighbouring valleys are reduced in relative terms. This is expected, as the model allocates summer grazing emissions to all grass categories including those common in hill areas, while housing, storage and manure spreading emissions are allocated only to good quality grassland. Furthermore, Figure 5.6.b also shows that dairy areas (e.g. Cheshire) have increased summer emissions to a greater degree than beef areas (e.g. Aberdeenshire), as some housing is still associated with dairy cows during the summer, i.e. the emission potential for dairy cows is greater than for beef cattle in summer.

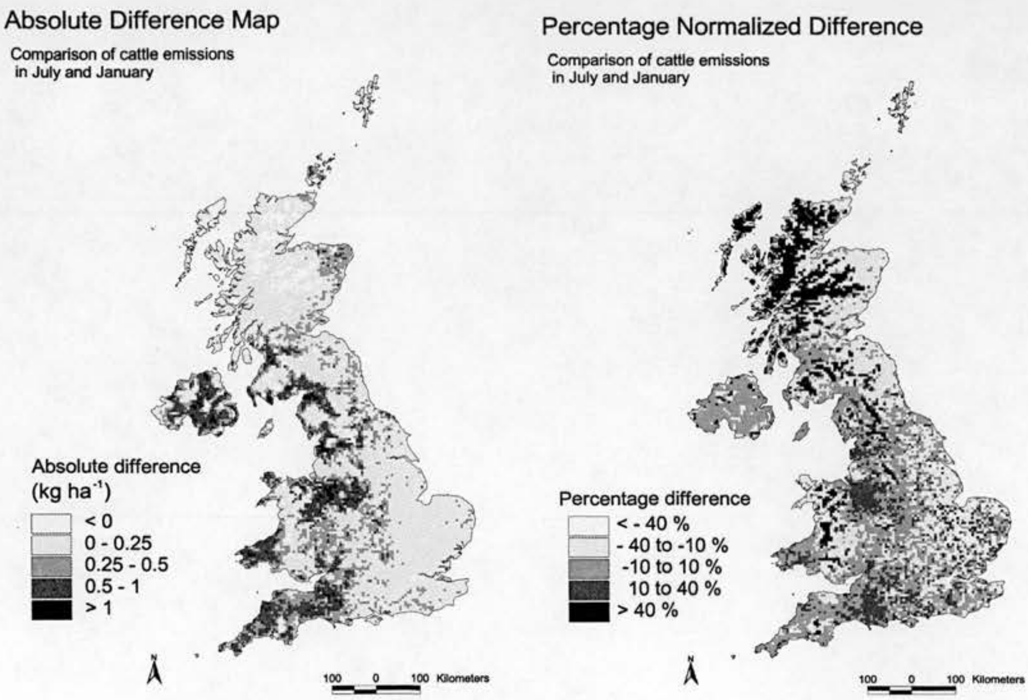

Figure 5.6. a) Absolute difference (kg ha^{-1}) in cattle emissions in summer compared with winter, i.e. January emissions minus July emissions. b) Percentage of normalized difference in cattle emissions in summer compared with winter, (i.e. normalized July emissions minus January emissions divided by January emissions).

The monthly activity data are highly generalised and are associated with large uncertainties. A major limitation is that they only incorporate agricultural practice, and therefore fail to take environmental factors such as temperature into account. These temporal uncertainties occur in addition to the traditional spatial uncertainties discussed in Chapter 3, and are linked to the location, rate, magnitude and timing of change. Future work should concentrate on improving the temporal activity data in order to reduce some of these uncertainties.

A limitation of the current monthly AENEID model is that it applies average temporal emission source strength evenly across the whole of the UK. Temporal emission source strength vary across the country, and this is particularly important regarding the length of the grazing season, which is generally longer in the south of the UK. One way to reduce this uncertainty is to model the length of the grazing season for each grid square in the UK and adapt the emission potentials accordingly. This is discussed further in Chapter 6.

5.7.6 Evaluation of the monthly AENEID model

The monthly emission results were compared with measured NH_3 concentration data to assess the robustness of the monthly emission estimates. Monthly measured concentration data were provided from the UK National Ammonia Monitoring Network (NAMN) (Sutton et al., 2001; Tang and Sutton, 2004). One of the aims of the NAMN is to assess temporal trends of concentrations, both intra-annual trends and inter-annual trends.

The magnitude of ammonia concentrations is primarily driven by NH_3 emissions, and therefore, although there are other factors which affect NH_3 concentrations, it is still informative to compare the modelled emission trend with the measured concentrations. This can give useful indication of the general seasonal emission pattern according to different emission sectors.

Measured NH_3 concentrations were available for 83 sites across the UK, which were assigned into four different groups (Sutton et al. 2001; Tang and Sutton, 2004) depending on the dominant NH_3 source in the area (cattle, pigs & poultry, sheep or background emissions). The average monthly NH_3–N concentrations (μg N m^{-3}) for the selected sites, aggregated by the dominant source types are shown in Figure 5.7, together with the seasonal trend in modelled NH_3-N emissions for the corresponding 5-km grid squares, which were derived from the monthly emission maps calculated in this study.

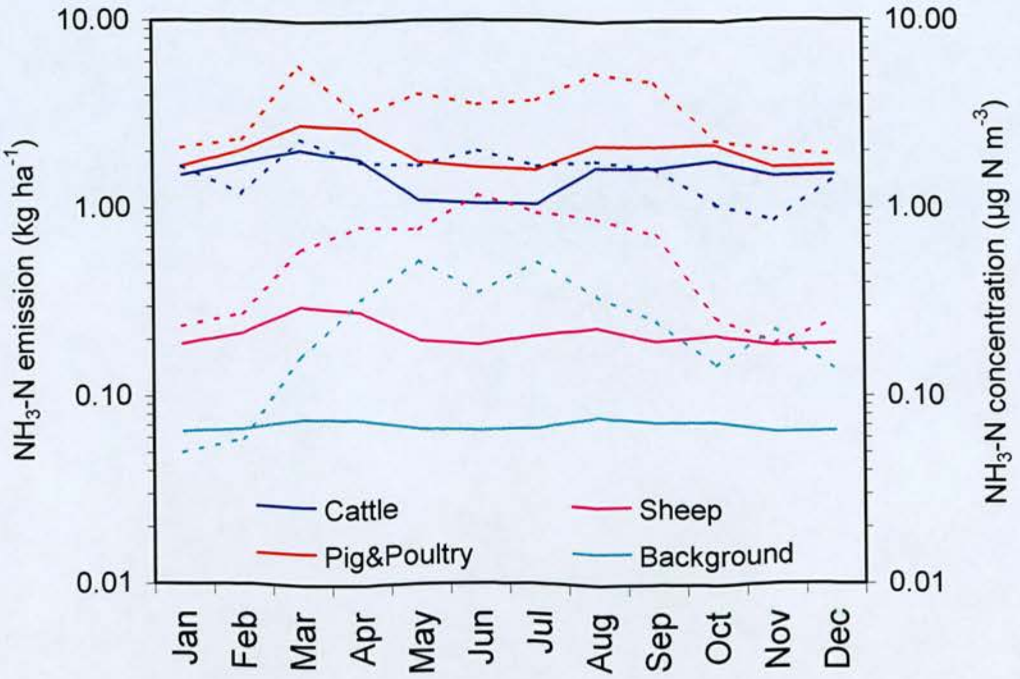

Figure 5.7. Average modelled NH_3-N emissions (kg N ha^{-1}) (——) and measured NH_3-N concentration (µg N m^{-3}) (-------) for the year 2000 at 83 sites across the UK in areas dominated by either cattle, pig & poultry, sheep or background emissions (logarithmic scale). NH_3-N concentration values derived from the UK National Ammonia Monitoring Network (Sutton et al., 2001).

From Figure 5.7, it is clear that the modelled temporal emission trend is similar to the trend in NH_3 concentrations for pig and poultry dominated areas, with high values in spring and autumn, and smaller values in summer and winter. For cattle dominated areas, the modelled emission trend shows low values in summer, but this is not as evident in the ammonia concentration values. This is expected to be due to environmental factors such as temperature, which have not been accounted for in the current monthly emission estimate. The temperature effect is also a likely explanation for the increased concentrations in sheep dominated areas and background emission areas during the summer, while the modelled emissions for these areas are more constant throughout the year.

The seasonal AENEID model developed in this thesis was used to derive spatio-temporal patterns in NH_3 emissions in the UK following the 2001 outbreak of Foot and Mouth Disease, where seasonal NH_3 concentrations were derived from monthly AENEID output and compared with measurements (Sutton *et al.*, 2006). The comparison suggests that modelled monthly NH_3 concentrations are larger than the measured concentrations and that the spring and autumn peaks were reproduced by the model, but not the winter peaks in NH_3 concentrations in the measurements. This result does not necessary imply an overestimation of modelled NH_3 emissions, but could be due to the specific location of sampling points in the modelled grid-squares and uncertainties in the dispersion model. Furthermore, the failure to pick up the winter peak implies that the seasonal emission model fails to incorporate manure spreading to frozen soils which occurs regularly in northern Britain (M.A. Sutton, CEH, pers. comm., 2005). The robustness of the monthly AENEID emission estimates should be further tested by generating concentration fields based on the monthly emission maps in atmospheric dispersion models (e.g. Fournier *et al.*, 2002), so that the maps can be adequately compared with the monitoring data of atmospheric ammonia concentrations.

5.8 Conclusions

Long-term measurements have shown that seasonal variations in measured NH_3 concentrations occur during the year. These variations have been associated with both climatic conditions (mainly temperature) and farming activity (such as manure application). Generally, ammonia emission inventories in the past have been calculated on an annual basis, therefore failing to capture these seasonal variations in emissions.

The temporal NH_3 emission approach developed in this study provides a general seasonal picture of ammonia emissions during the year. This information can potentially be applied to identify when monthly threshold levels of ammonia are exceeded and when abatement measures should be implemented. Furthermore, the seasonal NH_3 emission maps can be used as input to atmospheric transport models, which help to improve and interpret the seasonal dynamics in ammonia dispersion and deposition.

The calculated monthly emission maps showed a strong seasonal pattern, with the highest emissions during springtime and the lowest emissions during summer. This emission pattern reflects the temporal activity data, with cattle grazing outdoors during the summer, and most manure and fertilizer application occurring in springtime, with a smaller peak in autumn.

The modelled seasonal emission trend corresponds fairly well with measured NH_3 concentrations. The model should however be validated with measurements in greater detail by calculating monthly NH_3 concentration from the emission maps through application of atmospheric transport models.

Future studies should concentrate on reducing uncertainties in the temporal activity data, and to develop approaches to include environmental factors such as temperature. Furthermore, regional differences in the cattle grazing season in the UK should be incorporated, as the cattle grazing season has been identified as a significant temporal uncertainty.

6 Assessing the impact of a variable dairy cattle grazing season in the AENEID model

6.1 Introduction

The length of the cattle grazing season has been identified as the most sensitive input parameter to an ammonia emissions inventory (Webb and Misselbrook, 2004). This is because housing and associated storage and spreading emissions are substantially greater than grazing emissions as a fraction of the nitrogen excreted. The length of the grazing season varies considerably within the UK due to variations in climatic factors, with the general pattern being a longer season in the south and west, declining with increasing latitude and altitude. It is therefore important to take the length of the grazing season into account in a spatial context when modelling NH_3 emissions. There may also be variations in the length of the grazing season between years due to weather. Farming practice and type of cattle also influences the length of time that cattle spend outdoors grazing. For instance, despite cold conditions in highland areas in winter, some beef cattle are still outdoors all year. Dairy cattle tend to be more dependent on the grazing season than beef cattle, hence this study is narrowed down to assess the impact of a variable cattle grazing season for dairy cattle only.

The ammonia emission potential from housed cattle is significantly greater than from animals grazing outdoors, as the excreta quickly infiltrates the soil during grazing, while housed livestock are associated with housing, storage and spreading emissions. As part of the task of modelling ammonia emissions in the UK it is therefore important to estimate the period livestock spend outdoors grazing and the period of the year that they are housed to adjust for regional variations in the emission potential for cattle, due to variations in the cattle grazing season. Hence, in this chapter, the impact of applying a variable dairy cattle grazing season in the AENEID model is assessed, with respect to spatial and inter-annual variations. For this purpose, the length of the cattle grazing season in the UK was modelled.

6.2 Factors affecting grass growth

The grazing season is strongly dependent on the grass growing season, which in turn depends on climatic factors, of which temperature is the most important. Rainfall can also be important, since drought conditions can be a limitation to grass growth during a warm, dry summer. Rainfall is however not considered to be a major limiting factor to the start of the growing season, because soil water content in the UK is normally at its maximum in springtime. However, rainfall will also limit the grazing season, as saturated soils are more sensitive to poaching, i.e. soil compaction and destruction of the sward (Smith, 1976; Topp and Doyle, 1996).

Grass growth is also affected by solar radiation, especially the rate of growth, as longer days (more solar radiation) is beneficial for growth (Briggs and Courtney, 1991). South facing slopes are more favourable for growth than north facing ones, where direct sunshine in morning and evening is cut off by shade, and the length of time exposed to sunshine is therefore shorter (Smith, 1976). Altitude and aspect both affect growth, and as temperature decreases with altitude, so does the growing season. In Scotland, the growing season starts on average 4.3 days later for every rise in altitude of 100 m (Grigg, 1995).

Other factors affecting the start of the grazing/growing season are type of grass species grown, and the timing of the end of the previous grazing season (Mayne, 2001). Soil characteristics may also be important. Taylor (1967) noted that different soil characteristics can result in differences in the length of the growing season of about ten days to two weeks in the same area, where sandy soils warm up more quickly than compacted, wetter soils.

6.3 Growing season definitions

The terms growing and grazing season are distinct from each other and should not be used synonymously. For example, as enough grass has to be on offer for the animals before grazing can commence, while some grass growth may continue beyond normal grazing periods. Many definitions of the growing season can be found in literature; most of them are in some way associated with temperature.

6.3.1 Base value approach

The most common way to define the growing season in the UK is using a threshold value which corresponds to the onset of plant growth. This "base-value approach" assumes that growth takes place during the period when the soil or air temperature is above a certain base value. Temperature values from the immediate environment of the plants, e.g. soil temperatures, are probably more accurate to use, but air temperature data are in many cases more readily available. Different grass species will respond differently to the threshold value.

A threshold of 6 °C appears to be the most common air temperature value used to define the growing season (Gregory, 1954; Taylor, 1967; Smith, 1976; House, 1982; Briggs and Courtney, 1991). Other growth thresholds have also been applied, e.g. 5 °C (Mitchell and Hulme, 2002) and 5.6 °C Grigg (1995). Many different threshold values also exist for soil temperatures at different depth. Frame (1992) suggests that grass growth takes place when soil temperature at 10 cm depth is 5-6 °C. Smith (1976) suggests a threshold value of 6 °C at as much as 30 cm soil depth.

Broad and Hough (1993) argued that "the base-value definition" does not clarify what is meant by growth, since grass can grow at temperatures near freezing point. Instead they used an alternative way of calculating the growing season based on growth rates calculated at weekly intervals. A threshold value of 5 kg ha^{-1} d^{-1} (dry matter) was used to define the growing season. When comparing the results with the 6 °C (air temperature) base value approach described above (also defined to represent the end of the growing season), they found that their method resulted in a shorter growing season. The commonly accepted threshold value of 6 °C was not equivalent to the mean temperatures at the start and the end of the growing season based on growth rate. Instead, the new definition based on grass growth rates corresponded to 5.1 °C (4.5 °C – 5.5 °C) for the start, and 8.3 °C (7.9 °C – 8.8 °C) for the end of the growing season. Broad and Hough (1993) therefore concluded that 5 and 8°C are more suitable temperatures for the definition of the growing season than the more commonly used threshold of 6 °C.

6.3.2 Rainfall based definition

Although temperature-based definitions for the growing season are most common, rainfall may also play an important role in grass growth. Extremes of wetness and drought are, however, more important for the growth rate during the growing season, and not so much a limitation to the length of the growing season.

Hurst *et al.* (1967) calculated the number of grass growing days in England and Wales based on monthly rainfall and monthly potential transpiration in 1962. In this context, grass growing days were defined as *"the number of days between April and September (inclusive) when soil moisture deficit did not exceed 2 in"*. The resulting map not only represented the number of grass growing days, but also the dryness or wetness of a summer. The fewest grass growing days occurred in the south east of the UK, increasing to the north west, as shown in Figure 6.1. This pattern differs from growing season maps based on temperature, where the growing season is longest in the south and west declining with increasing latitude and altitude.

Figure 6.1 . Grass growing days in England and Wales during April to September, 1962, based on monthly rainfall and potential transpiration (Hurst and Smith, 1967).

6.3.3 Accumulated temperatures

Another temperature-based approach to define the start of the growing season that became particularly popular in the 1990's is the use of accumulated temperatures. Well established concepts are 'growing degree days' (common in the US), and 'T-Sum 200' (used in the NL and the UK). Both methods are based on accumulated temperatures and build on the concept that plant development only occurs above a threshold temperature value that varies with type of plant species.

Gregory (1954) produced maps of accumulated temperatures above 6 °C based on mean monthly temperatures, as shown in Figure 6.2. This map gives an indication of the intensity of growth rather than the start and end of the growing season. Since the map is based on monthly instead of daily mean values, daily and weekly variations are lost. The map clearly shows the reduction in growing period with increasing altitude and latitude, as well as an increase in growing period in the oceanic south west due to mild winters in these areas.

Figure 6.2. Mean annual values of accumulated temperatures, (Gregory, 1954).

In the UK, the 'T-Sum 200' method is based on accumulated temperatures to assist farmers in timing initial fertilization with the start of grass growth (Frame, 1992). The method was originally developed in the Netherlands, but has been applied in the UK since the mid 1990's. The average daily temperatures are cumulatively summed from the first of January, excluding any negative values. When the accumulated total of the daily averages reaches 200 ^{o}C, fertilizer N should be applied (provided that ground and weather conditions are satisfactory for spreading fertilizers).

It may be argued that the T-Sum 200 method is unsuitable for the UK since it originally was developed in the Netherlands, where flat coastal land dominates. IGER (2000) has investigated the T-Sum 200 method in mountainous areas at Bronydd Mawr (305 m) in Wales. In the year 2000, T-Sum 200 was reached on the 18th of February, but the soil did not reach 5.5 ^{o}C until three weeks later (the 11th of March). This study therefore suggested that the T-Sum 200 methodology may not be as accurate in mountainous areas.

Farmers Weekly and Farmers Weekly Interactive (FWi) give weekly updates on the T-Sum 200 in the UK (www.fwi.co.uk). The results are presented on a 50 x 50 km grid map, as shown in Figure 6.3. Predicted dates, or T-Sum arrival dates, are presented for each grid square and can be found in a separate table on the website. These dates identify when the T-Sum threshold of 200 is predicted to be reached. The south west part of the UK is generally the first area to reach the threshold. Due to the large size of each grid square, major local variations can occur within each square.

6.4 Grazing Season definitions

The grazing season is strongly dependent on the growing season, as there has to be enough grass available for grazing. However, the grazing season is also dependent on agricultural practice and those factors that farmers consider important for determining the start and end of the grazing season, e.g. common practice, hay supply, stocking rates etc.

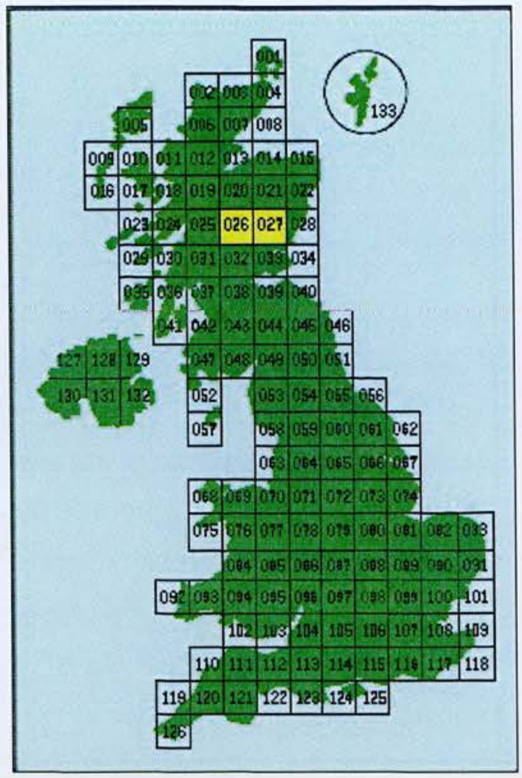

Figure 6.3. Grid ids for the T-Sum 200 Map published on FWi. The "T-Sum arrival date" (or predicted arrival date) for each grid square can be found in a table, also published at FWi. (after FWi – www.fwi.co.uk)

The start of the grazing season can be defined as the delay after the start of the growing season, to allow for enough grass to develop. The initial growth tends to be rapid and reaches its peak within 6 to 8 weeks (Briggs and Courtney, 1991; Frame, 1992), as shown in Figure 6.4. Frame (1992) suggests that the grazing season starts 5 to 6 weeks after the start of the growing season, and/or three to four weeks after fertilization, in order for enough biomass to develop for grazing.

The end of the grazing season is more difficult to determine than the start, since the grazing season does not necessarily end when the grass stops growing. Topp and Doyle (1996) list four criteria for when the grazing season is considered to end:

- The metabolizable energy available from the grass intake does not meet the metabolizable energy requirements of the livestock
- The predicted grass intake falls to less than 20 % of the potential level
- The soil has returned to field capacity for five consecutive days (poaching is likely to occur)
- The growing season has ended

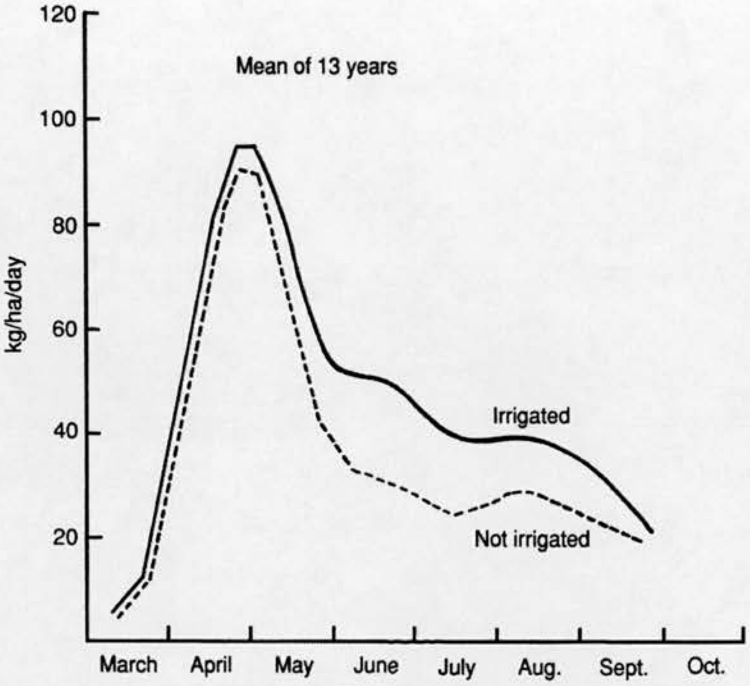

Figure 6.4. Average seasonal grass growth rates in the UK (from Frame, 1992)

The first two criteria are difficult to include in a model as they are not based on easily quantifiable parameters. The latter two criteria can however be included, based on temperature and rainfall data.

Smith (1976) and Topp and Doyle (1996) similarly suggest that the end of the grazing season can be considered to coincide with the date at which the soil moisture returns to field capacity, i.e. when the available soil moisture is greater than or equal to the available water capacity of the soil. Smith (1976) suggests that average summer rainfall is the best factor for predicting the date when the soil moisture returns to field capacity, since soil moisture balance parameters are best expressed in terms of rainfall. The grazing season generally ends later in the south and east and earlier in the north and west due to the larger annual rainfall in the north and west leading to poaching damage to the soil (see Figure 6.1).

6.5 Modelling the grazing season

For the purpose of modelling ammonia emissions on a national scale, estimating the length of the grazing season based on a temperature approach was considered to be sufficient to give a broad indication of the variability of the grazing season within the country and between years. Rainfall and transpiration data were not included in the present model, despite the fact that soil water content may potentially act as a constraint to the start and end of the grazing season, as grazing is not possible when the soil is too moist. Furthermore, temperature and rainfall are implicitly linked, i.e. high temperatures are likely to be associated with greater transpiration, and therefore drier conditions. Temperature therefore indirectly accounts for poaching effects in the model to some degree. In a local (fine scale) study it may be appropriate to collect and include the interaction of local environmental conditions and to include a poaching constraint, but for the purpose of assessing the impact of a variable grazing season in the AENEID model, the method applied for estimating the start and end of the growing season can be less detailed.

6.5.1 Start of the grazing season

Davies (1986) compared the use of a base temperature and accumulated temperatures and could not find any significant correlation between the dates for soil 5.5 °C and air T-Sum 200. During the period between 1967 and 1986 the soil temperature of 5.5 °C was obtained on average 27 days after the air T-Sum 200. For the purpose of modelling the start of the grazing season in this study, T-Sum 200 was chosen, as this reflected the fact that this method is already widely used in agriculture (for predicting timing of spring fertilization) and it is therefore more likely to reflect current agricultural practice.

It may be recalled (Section 6.3.3) that the T-Sum 200 method may be more uncertain at high altitudes, since the method was developed in the Netherlands for flat coastal land. However, for the purpose of this study, the methodology is considered to be acceptable despite the possible errors in high altitude areas, as it still reflects current agricultural practice. Furthermore, the emission potential for cattle is only adjusted for dairy cattle, and dairy farming tends to occur mainly in lowland areas (Figure 6.9.a).

Based on the information reviewed above, the start of the grazing season is calculated here 6 weeks after the date of T-Sum 200, in agreement with Frame (1992) and Webb *et al.* (2005).

6.5.2 End of the grazing season

For the purpose of the present model, the grazing season is assumed to end when the grass is estimated to stop growing according to an autumn base temperature. This contrasts with Webb *et al.* (2005) who assumed that the grazing season ends on the 16th of October, without taking any account of climatic variations. A base value of 8 °C was used in the present model, reflecting comparison of grass growing rate and temperature by Broad and Hough (1993), who concluded that 8 °C is a more suitable temperature to define the end of the grass growing season than 6 °C. Since the study of Broad and Hough was based on the biomass production (growth rates) of grass, this conclusion is considered as being more reliable for the definition of autumn grass growth than 6 °C.

6.5.3 Modelling approach

The modelling approach chosen to represent the start and the end of the grazing season in this study is therefore based on the following assumptions:

Start – *it is assumed that the grazing season starts six weeks after T-Sum 200 of the air has been reached.*

End – *the grazing season is assumed to stop when the average air temperature for 5 consecutive days has decreased to 8 °C.*

In addition, threshold dates for the start and end of the grazing season were set to avoid unrealistic dates, e.g. an all year grazing season, or an extremely early or late start/end to the grazing season. These constraints were set as follows:

Start - From the 16th of March until the 15th of May.

End - From the 16th of September, until the 15th of November.

These constraints were based on the assumption by Webb *et al.* (2005), where the grazing season is assumed to end on the 16th of October, as farmers typically bring in their dairy cattle in the middle of October. Furthermore, an average 180 days grazing season in the UK has been widely used (Misselbrook *et al.*, 2004; Webb *et al.*, 2005), which indicates that the start of the grazing season should occur around the middle of April. The start and end of the grazing season were allowed to fluctuate in the model by + / - one month between these two dates, i.e. setting grazing season limits of 124 to 244 days.

The temperature-based modelling approach requires spatially distributed daily air temperature data to define both the start and the end of the grazing season. The initial intention was to use fine scale meteorological data at 1-km grid resolution, but this was not possible due to lack of availability. The finest resolution for temperature data available for this study was at 5 x 5 km, but only as monthly averages (Naomi Eastment, Met Office, pers. comm., 2005). Daily temperature therefore had to be interpolated from the monthly means.

The maps representing the start and end of the grazing season were calculated based on the temperature data specified. The length of the grazing season was then calculated as the difference between the two maps for each 5 x 5 km grid cell. If no temperature data were available, that grid cell was allocated the average UK value for the length of the grazing season.

6.5.4 Calculation of new emission potentials

The regional variation in the length of the grazing season (5-km resolution) in the NH_3 emission model was implemented in the AENEID model by weighting the emission potential according to number of grazing days per grid cell. This was achieved by calculating two different emission factors for dairy cattle, one emission factor representing the grazing season (EF_{gz}), and the other representing the housing season (EF_{ho}). The final dairy cattle emission factor for each 5 x 5 km grid cell was then calculated according to Equation 6.1.

$$EF_{annual} = \text{grazing season}_{(d)} * EF_{gz} + \text{housing season}_{(d)} * EF_{ho} \qquad (6.1)$$

EF_{annual} (kg NH_3-N animal^{-1} yr^{-1})
EF_{gz}, EF_{ho} (kg NH_3-N animal^{-1} d^{-1})

Emission factors for a dairy cow during the housing season and the grazing season (kg NH_3-N animal^{-1} d^{-1}) are shown in Table 6.1. Although grazing animals do not produce any manure to be stored, the emission potential for grazing cattle also includes the emission potential for manure storage, as manure is assumed to be stored all year round. (Note that the grazing emission potential in Table 6.1 has not been derived for cattle that are grazing all year round, but should only be applied for cattle that are grazing for part of the year.)

Table 6.1. Cattle emission factors for a grazing dairy cow during the grazing season and housing season, compared with the emission factor applied by Misselbrook *et al.* (2004).

Cattle category	**Grazing season** [a)] Emission factors (kg NH_3-N animal^{-1} d^{-1})	**Housing season** [b)] Emission factors (kg NH_3-N animal^{-1} d^{-1})	**Annual** [c)] Emission factors (kg NH_3-N animal^{-1} d^{-1})
Diary cows & heifers	0.0407	0.0924	0.0655
Dairy heifers in calf	0.0251	0.0482	0.0362
Average Dairy	0.0329	0.0703	0.0508

[a)] Includes grazing and storage emissions
[b)] Includes housing, storage and spreading emissions
[c)] As assumed by Misselbrook *et al.* (2004), i.e. based on a grazing season of 190 days.

The emission potentials presented in Table 6.1 are based on the following assumptions:

- Grazing emissions are assumed to occur only during the grazing season. Therefore the total grazing emission derived from IAEUK was divided by the number of days of grazing assumed in IAEUK (190 days) and multiplied by 365 days, in order to calculate EF_{gz}.

- Housing emissions are assumed to occur both during the grazing season and the housing season. Although dairy cattle are outdoors grazing during the grazing season, housing emissions continue for two reasons. Firstly, dairy cattle still spend part of the day indoors for milking (Phillips *et al.*, 1998), and secondly livestock excreta have a long-term potential for NH_3 losses unless the NH_4^+ is bound or converted (Erisman and Monteny, 1998).

- Storage emissions occur both during the grazing and the housing season, as livestock manure is stored all year round. The emission potential for storage was therefore assumed to be the same during the grazing season as during the housing season.

- Spreading emissions are assumed to be proportional to the amount of manure produced, hence spreading emissions should be calculated based on the length of the housing season. The emission potential for spreading was therefore calculated as the total spreading emission divided by number of days housed (175 days) and multiplied by 365 days.

6.6 Results and Discussion

6.6.1 Cattle grazing season 1990 - 2000

The average length of the cattle grazing season during the period from 1990 to 2000 was modelled to estimate the likely variations in the cattle grazing season in the UK between years, but also to assess spatial variation (Figure 6.5). The average modelled cattle grazing season for the UK during this period varied from 179 days (in 1991) to 220 days (in 1990), i.e. a potential variation of more than a month (41 days). These results suggest that the potential grazing season varies significantly from year to year, and consequently the emission potential for dairy cattle changes between years. For instance, the emission potential for dairy cattle is 8.8 % greater in 1991 compared with 1990, due to a shorter cattle grazing season. However, as previously mentioned, the length of the grazing season is also dependent on agricultural practice and farmers choice, hence the potential season may not be utilised fully.

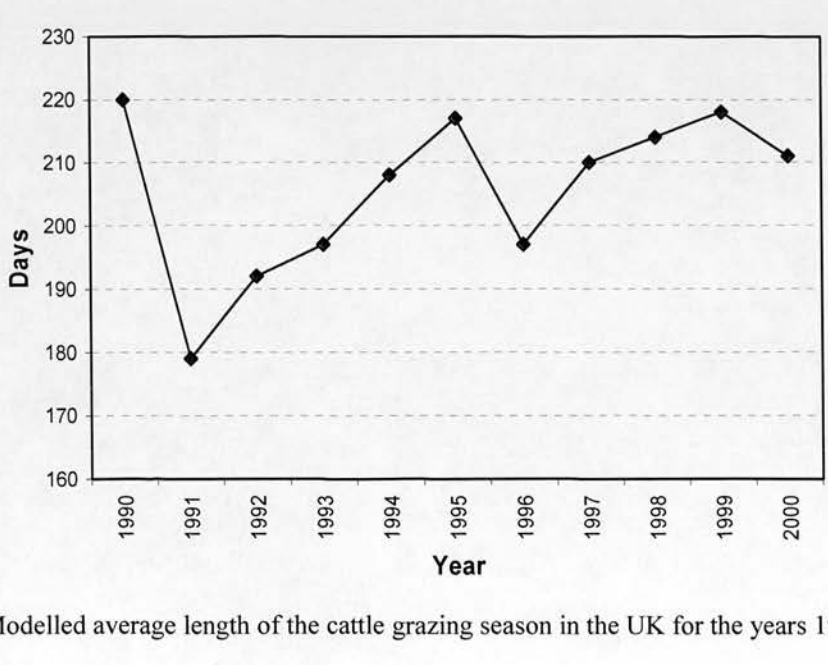

Figure 6.5. Modelled average length of the cattle grazing season in the UK for the years 1990 – 2000.

The average grazing season modelled here is slightly longer than the grazing season for dairy cattle of 190 days assumed in the IAEUK. This may indicate that the length of the grazing season applied in the IAEUK is too short. However, uncertainties in the modelling parameters and constraints will have to be assessed further to validate this statement. The current modelling approach is based on average monthly temperature values, from which daily temperature values were interpolated. If daily temperature values had been available, it is likely that the modelled grazing season would have been shorter, as the monthly means fail to capture daily fluctuations in temperature. For instance, the T-Sum200 method only adds daily temperature values above 0 °C. It is likely that the daily fluctuations of temperatures at the start of the year (when it is likely that daily mean temperatures may be below 0 °C) may be lost when applying monthly averages. A similar problem occurs for the ending date of the grazing season, as the interpolated daily values decrease gradually, and daily variations are lost. Hence the condition of 5 consecutive days with a mean temperature value of below 8 °C may occur sooner (or later) than when monthly temperature values are applied.

6.6.2 Grazing season maps for 1990, 1996 and 2000

The modelled length of the grazing season for the years 1990, 1996 and 2000 in a spatial context is shown in Figure 6.6.a-b and Figure 6.7.a. All three maps show a varying length of the grazing season from 124 to 244 days (set by the constraints).

It can be seen that the grazing season is longest in lowland areas in the south and west, and significantly shorter in upland areas. The longest grazing season is modelled for 1990, and the shortest for 1996, and the difference between these two years is shown in Figure 6.7.b to provide an indication of how much the grazing season may vary between years. Although the average length of the grazing season differs by 23 days between 1990 and 1996, Figure 6.7.b suggests that the grazing season may vary by as much as 2 months (60 days) between years in some areas of the UK.

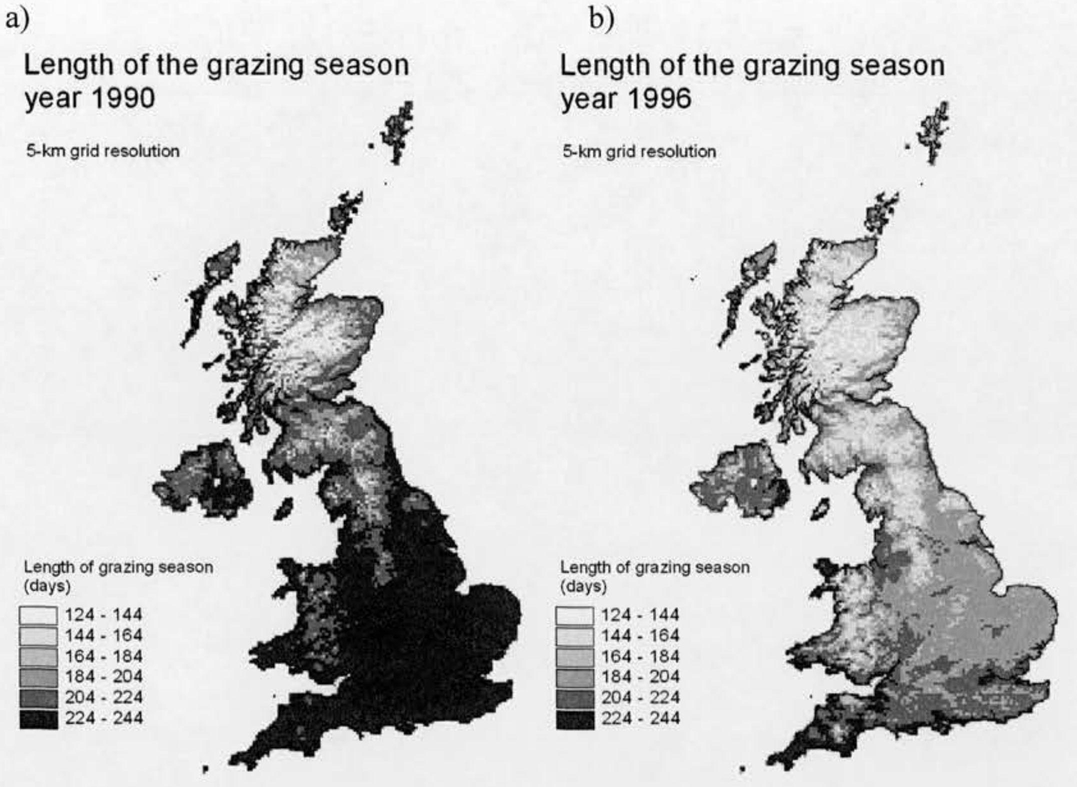

Figure 6.6.a-b. Modelled length of the grazing season (days) for a) year 1990, and b) year 1996

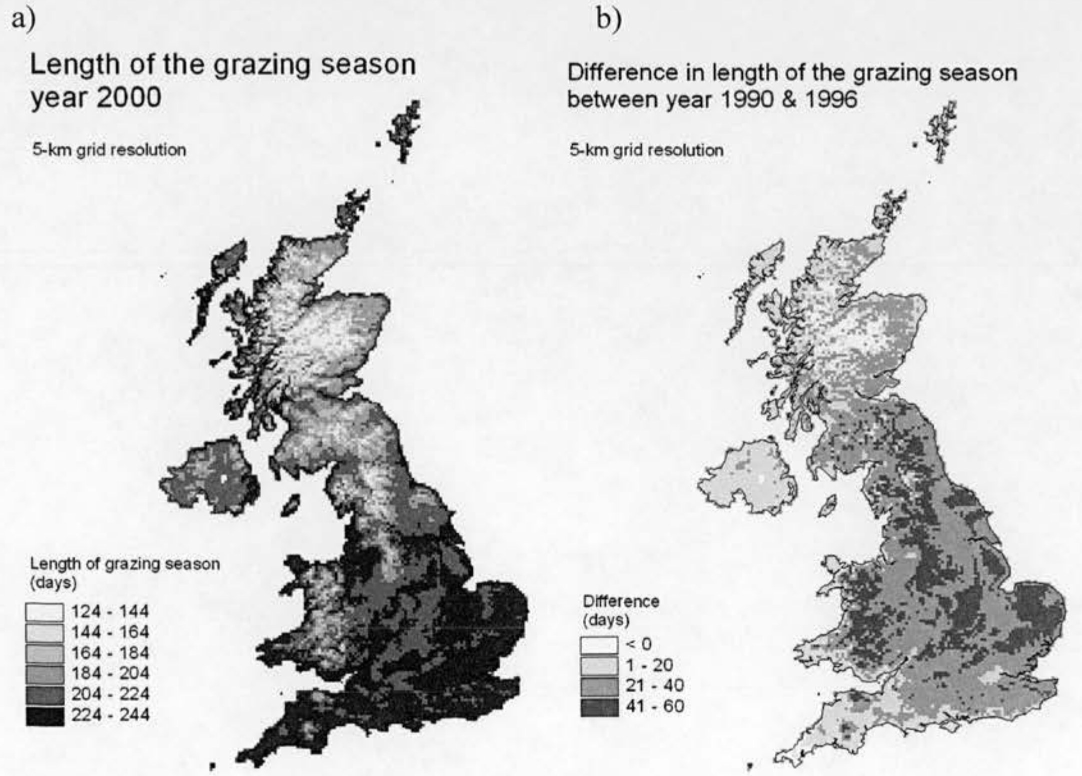

Figure 6.7.a-b. Modelled length of the grazing season (days) for a) year 2000. b) Difference in days between year 1990 and 1996.

A quantitative comparison between the three years is provided in Table 6.2. The average length of the grazing season in the UK for the three years varies from 197 days (year 1996) to 220 days (1990). The average starting date varies between the 24th of March (1990) to the 17th of April (1996). The average end date for the three years is fairly constant (30th of October in 1990, 28th of October in 1996, and 27th of October in 2000).

Table 6.2. Average starting date, ending date and length of the grazing season. % of grid cells falling outside the date constraints are also shown.

	1990	1996	2000
Average length of the grazing season	220 days	197 days	211 days
Average starting date	24th of March	17th of April	30th of March
Average ending date	30th of October	28th of October	27th of October
% grid cells with start date $<$ 16th March	24.4 %	0.9 %	2.4 %
% grid cells with start date $>$ 15th May	1.1 %	12.1 %	2.3 %
% grid cells with end date $<$ 16th September	2.7 %	0.4 %	0.4 %
% grid cells with end date $>$ 15th November	3.0 %	1.1 %	9.5 %

The constraints in the modelling approach for the start and end of the grazing season were assessed by calculating the percentage of 5 x 5 km grid cells that fall outside these time limits (Table 6.2). The results suggest that the constraints set for the start of the grazing season are appropriate, as the percentages of grid cells that fall before and after the constraint dates are very similar and small for year 2000, which is a more representative year than the more extreme 1990 and 1996 estimates. However the significant number of grid cells (24 %) falling outside the start date constraint (16th of March) for 1990, indicates that the model fails to capture an earlier start to the grazing season due to a warm early spring. The end of the grazing season constraints do not account for as much of the variability encountered in the modelled output as the start (with the exception of year 2000), but can still be regarded as appropriate as the percentage outside the constraints is still low.

Figure 6.8 shows the spatial variability of the start (1990) and end (2000) of the grazing season within the UK. For these years, a significant proportion of the constraint dates were hit due to a warm spring (1990), and a cold autumn (2000), which points to a limitation of the T-Sum200 limit for year 1990, and the 8 °C limit for year 2000. The significant spatial variability of the start and end of the grazing season within the UK indicates that the constraints should maybe be set wider for some parts of the UK.

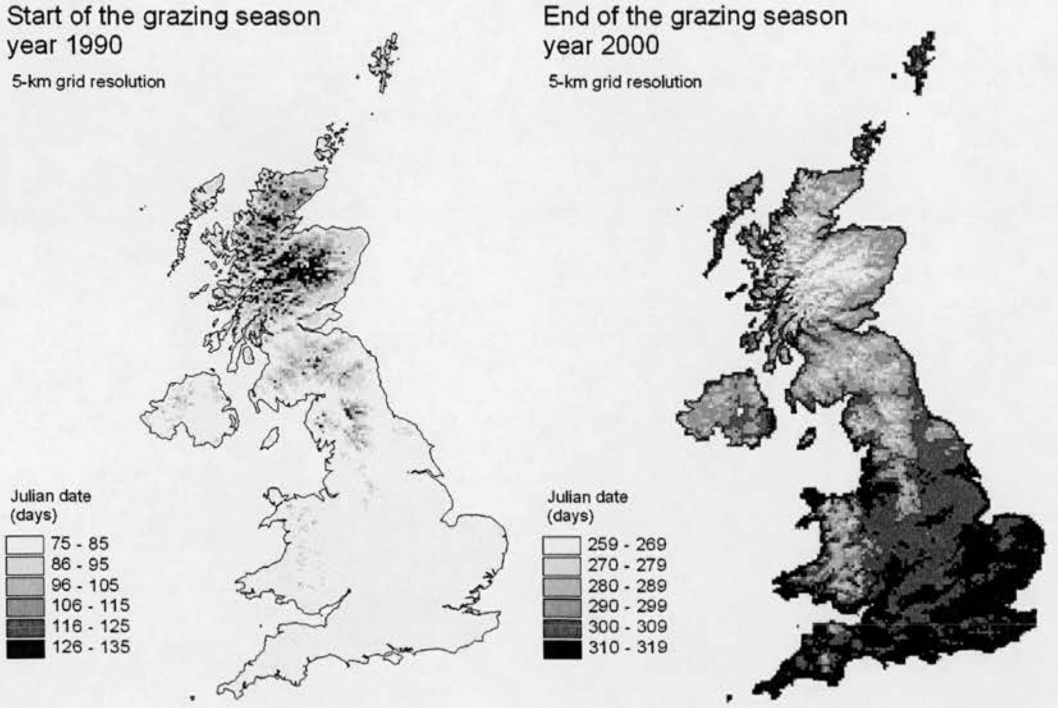

Figure 6.8. a) Start of the grazing season 1990 (julian date), b) end of the grazing season 2000 (julian date).

6.6.3 Dairy vs. beef dominated areas

The average length of the grazing season was calculated for areas dominated by dairy and beef respectively, to assess the potential differences between these two cattle groups (Figure 6.9). Dairy and beef dominated areas were derived from a dominance source map (Figure 8.9), which showed that dairy farming dominate in lowland regions, while beef farming is more common in upland areas. Grid cells with a small number of cattle (where the cattle grazing emission < 2.6 kg ha^{-1} yr^{-1}) were omitted from the assessment to avoid allocating grid cells to either beef or dairy based on only a small number of cattle. This resulted in only 90 % of cattle being included in the analysis. Table 6.3 shows the average length of the cattle grazing season in dairy and beef dominated areas, and the results suggest that the cattle grazing season is about 10-15 days longer in dairy dominated areas compared with beef areas. Dairy cattle tend to be kept on higher quality grassland than beef and in less marginal areas, which is also reflected in the grazing season, exemplified by a longer grazing season in lowland areas with good quality grassland.

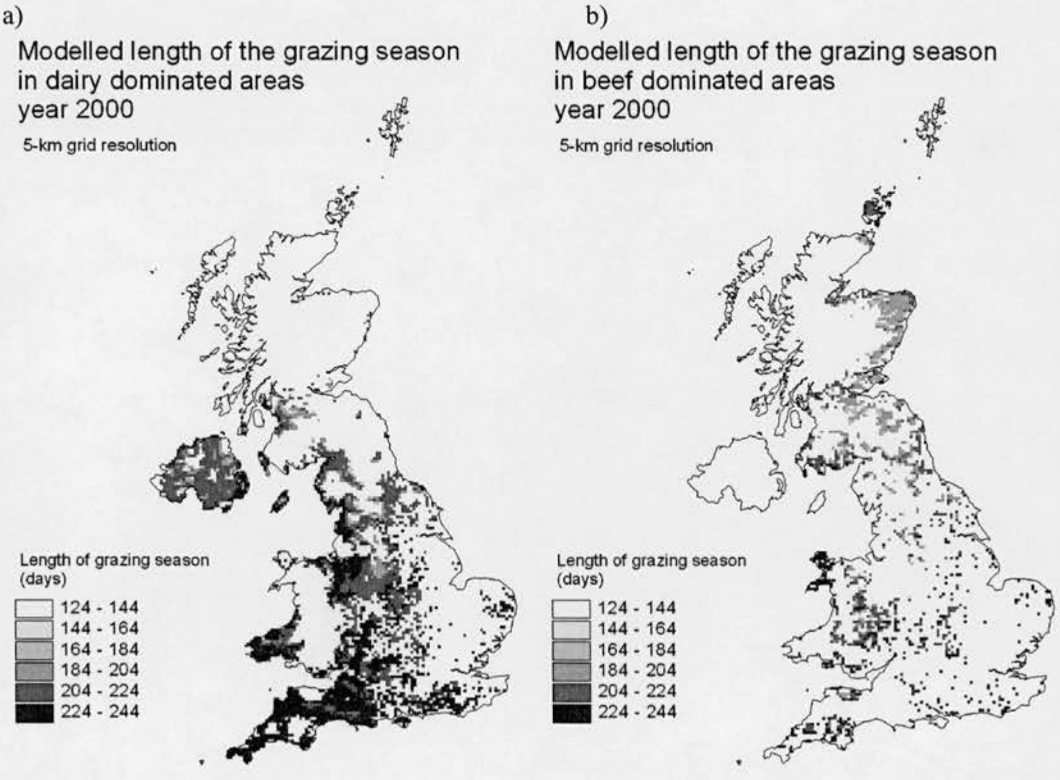

Figure 6.9. Modelled length of the cattle grazing season in a) dairy dominated, and b) beef dominated areas. (Note, grid cells with a small number of cattle were omitted, hence only 90 % of cattle were included)

Table 6.3. Average modelled length of the grazing season in dairy dominated and beef dominated areas

Year	Dairy dominated areas	Beef dominated areas
1990	232 days	222 days
1996	206 days	191 days
2000	223 days	211 days

6.6.4 Total dairy cattle emissions

Total dairy cattle emissions (NARSES categories n1-n3) were calculated for the UK for all three years based on the annual emission potential (IAEUK) and compared with the emission when the dairy cattle emission had been modified (derived from Equation 6.1 and Table 6.1) to incorporate variations in the grazing season. Table 6.4 shows that overall the modified approach results in lower emissions, due to a longer grazing season applied for all three years compared with the IAEUK (190 days). The greatest difference between the two approaches occurs for 1990, which is characterised by a long grazing season. This suggests that uncertainties in the NH_3 emission inventory, due to variation in the grazing season, could be reduced if the length of the cattle grazing season is incorporated in the inventory (provided that the uncertainties in the modelled grazing seasons are small). Therefore, further work should concentrate on reducing uncertainties in the modelling approach, taking into account agricultural practice and its relationship to the growing season. In addition to these mean countrywide differences, the approach allowing a variable grazing season will also lead to local differences in calculated ammonia emissions which will have impact on local ammonia concentrations and deposition.

Table 6.4. UK dairy cattle emission based on the original approach, and the new approach when the length of the cattle grazing season was incorporated.

	UK NH_3-N emission from Dairy cattle (kt)			
Year	IAEUK approach	New approach (grazing season incorporated)	Difference (kt)	Difference (%)
1990	69.0	62.9	6.1	8.8 %
1996	70.0	67.6	2.4	3.4 %
2000	63.0	58.7	4.3	6.8 %

6.6.5 Uncertainties in the model

When modelling the length of the grazing season, uncertainties are associated with the method applied, as well as in the quality of the input data. The lack of data on current agricultural practice is likely to be the largest source of uncertainty, as it is essential to know how farmers make their decision regarding when to put animals out to grass, and when to bring them back indoors for the winter.

For the present study, there was uncertainty associated with the temperature data, which were only available as monthly averages rather than daily values. Temperature may vary significantly on a local scale both spatially and temporally (Pielke *et al.*, 2002), and substantial variations are likely to occur within each 5 x 5 km grid cell. Further work should concentrate on obtaining and applying daily temperature values at a higher spatial and temporal resolution, to reduce uncertainties in the modelling approach associated with the input temperature data. Furthermore, the incorporation of rainfall data should be assessed, as this could potentially improve the modelling result, particularly for the end of the grazing season, which tends to occur earliest in areas with large annual rainfall.

The current approach to adjust for regional differences in the grazing season is only incorporated in the model with respect to the variability in emission potentials across the UK, hence variability in the location of emissions have not been incorporated (i.e. in relation to the spatial weightings of the AENEID model within parishes, Section 3.4). For instance, during the housing season, emissions tend to be located to those landcover types where the cattle houses are most likely to be, while emissions from cattle during the grazing season are also located to landcover types where grazing is assumed to occur. Consequently, in areas with longer grazing, cattle emissions would be more spread out onto grassland of all qualities and less to "good grass" and arable. It may therefore be argued that regional variations in the spatial location of emissions depending on whether cattle are housed or out grazing should be integrated into the AENEID model. At this stage of development, possible regional variations in the apportioning percentages affecting the spatial location of emissions have not been accounted for. These variations would have a particular impact in the monthly AENEID model (Chapter 5) for those spring months representing the transition from housing to grazing, and those months where the grazing season changes into the housing season. Further research should therefore focus on how to incorporate these regional variations in apportioning percentages across the country as a consequence of local variations in the length of the grazing season. A possible solution would be to calculate the average number of grazing days for each aggregation zone, and adjusting the apportioning percentages accordingly. Further work should also concentrate on collecting information on local agricultural practice regarding the start and end of the grazing season to validate the modelling results.

6.7 Summary and conclusions

The length of the grazing season has been identified as one of the most sensitive input parameters in the current NH_3 emission inventory. Dairy cattle are considered to be more dependent on good quality grassland compared with beef cattle, which can sometimes be outdoor all year around, depending on the type of beef cattle. This study therefore concentrated on modelling the length of the cattle grazing season for dairy cattle only. Due to the large differences in ammonia emission potentials from housed animals compared with grazing animals, the incorporation of the variability of the length of the grazing season would reduce uncertainties about the dairy cattle emission in the NH_3 emission inventory both within the UK and between years. A grazing dairy cow was estimated to emit 12.0 kg NH_3-N yr^{-1} during the grazing season compared with 25.7 kg NH_3-N yr^{-1} during the housing season (including associated storage and spreading emissions) in this study.

The grazing season is strongly dependent on the grass growing season, and is estimated to start on average 5-6 weeks after the grass starts to grow. In the UK, the growing season varies locally depending on climatic conditions, with temperature being the most important factor. Other factors include rainfall, transpiration, solar radiation, altitude, aspect, type of grass and previous grazing patterns.

Many definitions exist for the start of the growing season in terms of climatic variables. The most common definition used in the UK is the assumption that growth takes place when the average temperature is at least 6 °C. Another common method is the use of "T-Sum 200", where daily temperature values above 0 °C are summed from the beginning of the year, until the accumulated value has reached 200 °C. This method is widely used in the farming community to predict spring fertilization at the start of grass growth and was therefore chosen to determine the start of the growing season, as it is likely to reflect current agricultural practice. The grazing season is assumed to start six weeks after T-Sum 200 has occurred. The end of the grazing season is assumed to occur when grass growth ceases, i.e. when average daily temperatures during 5 consecutive days have decreased to 8 °C.

The length of the grazing season was modelled for the years 1990 to 2000, and results show variations by up to one month between years (corresponding to a difference in dairy cattle emission potential of 6.9 %). This suggests that the variability in cattle grazing season between years can influence the dairy cattle emission significantly.

The cattle grazing season was modelled at 5-km grid resolution for the years 1990, 1996 and 2000, with the emission potential for dairy cattle adjusted accordingly. The grazing season maps suggest that the cattle grazing season (and hence the dairy cattle emission potential) varies significantly within the UK, with generally a longer season (lower emission potential) in lowland areas in the south and west, and a shorter season (higher emission potential) in the uplands and in the north.

The dairy cattle emissions calculated for all three years were smaller when incorporating the modified emission potentials compared with the original AENEID approach, as the modelled grazing season was longer than the 190 days assumed in the IAEUK. This could suggest that the grazing season used in the IAEUK could be too short. However, the modelling approach and input data for calculating the length of the grazing season are associated with uncertainties that need further assessment to validate this conclusion. Further research should concentrate on incorporating higher quality temperature data (daily values with a higher spatial resolution) and potentially also rainfall data to reduce uncertainties. Furthermore, information on local agricultural practice regarding the start and end of the grazing season would be useful to verify the model.

7 Improving the spatial distribution of ammonia emissions from poultry farming in the UK

7.1 Introduction

The NH_3 emission maps based on the standard AENEID model show large "hot spots" of ammonia emission as a result of intensive poultry farming and, to some extent, pig farming. These source types are therefore of particular interest, as the most extreme localized environmental impacts of ammonia may result from poultry and pig farming. While the nature of these sources, i.e. large intensive livestock installations, suggests that this is likely to hold true in principle, the scale of the problem is probably overestimated by the current maps. The original AENEID model, described in Chapter 3, assumes that ammonia emissions occur close to the animals' estimated location, i.e. within the aggregation zone of origin. This may be true for some type of livestock, especially for land-based farming of cattle and sheep, but is less likely for intensive livestock units such as pig and poultry farms, especially when considering the fate of manure (storage and landspreading). The assumption that all manure spreading from pig and poultry farms occurs within the aggregation zone of origin was identified as one of the major uncertainties in the original AENEID model by Dragosits (1999). Intensive poultry farms are likely to produce manure in excess of amounts that can be utilised on suitable land near the farm, and therefore, some or all of the manure is transported elsewhere, either for landspreading or for incineration in purpose-built power stations.

In 2001 a Farm Practices Survey carried out in England indicated that over half of all poultry farms in England export some or all (on average about 92 %) of their poultry manure from their farm (Scott *et al.*, 2002). The extent to which manures are transported out of the parish of origin for application to fields or for use as a fuel source for electricity generation, means that the localized emissions (and potential environmental impacts) from poultry farms are overestimated in the current NH_3 emission maps. In the UK ammonia emission inventory 35 % of the poultry manure is assumed to be incinerated in power stations (Misselbrook *et al.*, 2004), but the remaining landspreading emissions are distributed evenly over all UK farms in AENEID. A second issue is that holdings registered for the June census may be located by head-office locations, rather than actual locations of the poultry houses. This will further add to the "hot spot" effect and apparent risk of localized impacts.

In order to improve the mapping methodology of the AENEID model, a case study was carried out, co-operating with the poultry industry to investigate uncertainties in the current mapping approaches and to develop a methodology to construct more reliable maps. Detailed data on a set of poultry farms in Scotland were collected and this information was used as the basis for the development of a module within AENEID for distributing poultry manure emissions over greater areas. A similar module was also developed for emissions from pig farms. The incorporation of these modules was expected to improve the inventory, primarily in areas with the highest NH_3 emission peaks.

7.2 Fate of poultry manure

About 4.4 million tonnes of poultry manure are produced annually in the UK (Smith *et al.*, 2001a). Instead of viewing this poultry litter as waste, it should be seen as a resource, providing energy, nutrients and organic matter. Main uses for poultry litter are: application to fields, composting, anaerobic digestion and direct combustion (Kelleher *et al.*, 2002).

Traditionally, poultry manure has been applied to grassland and crops, and Misselbrook *et al.* (2004) estimated the proportions at 47 and 53 % respectively. Storage of poultry manure in heaps may also lead to composting, with aerobic degradation of the biodegradable organic manure occuring during the storage period (Kelleher *et al.*, 2002). During the storage period nutrients are lost, for instance, ammonia volatilizes to the atmosphere, and nitrate may leach to water bodies. If the poultry manure is stored in anaerobic conditions, microbial organisms degrade the organic material, resulting in methane emissions and inorganic products (Kelleher *et al.*, 2002). The biogas produced in this process could be used as an energy source, and the sludge produced could be used as a fertilizer. The advantage of anaerobic storage conditions is that the volatilization rate of ammonia is much lower than for traditional storage.

7.3 Poultry manure – a case study

7.3.1 Background

The main objective was to develop an improved generalized methodology for the AENEID model to distribute emissions of ammonia from poultry farming. This included:

a) *Quantifying the fate of manures generated by example poultry farms (combustion and distances between farm and manure application).*
b) *Implementing a generalized model for the transfer of poultry litter and subsequent emissions outside of the parish in which the poultry housing occurs.*

The work was conducted in collaboration with the poultry industry to obtain data and information about several poultry facilities and the fate of the poultry litter from these facilities for four years between 1985 and 2005. Six poultry facilities were included in the study, three farm complexes (consisting of four to seven farms each), and three single farm units. A poultry farm may consist of many poultry buildings, and each poultry building consists of two poultry sheds. These poultry facilities were located in three out of the top four parishes with regard to poultry numbers in Scotland. This suggests that the poultry farms included in this study would give a representative approximation of intensive poultry farms in Scotland.

Mr. M. Stevens at Grampian Country Food Group Ltd. provided the detailed information about the poultry facilities. Grampian Country Food Group was assured that details of individual farms would not be disclosed in output datasets (because of animal security issues). For confidentiality reasons the names and detailed locations of the poultry facilities in this study can therefore not be shown. Conversely, it has been possible to show the main scientific findings in a way that does not break the confidentiality agreement.

7.3.2 Data input

All six poultry facilities in the case study were broiler units. One of the small farms was run as a broiler breeder unit until after 2002, when it was converted into a free-range broiler unit. At the same time, another of the small broiler farms was converted into a perchery barn unit for commercial layers.

All poultry houses selected for the study are ventilated by roof extraction (negative pressure) and have track feeders and nipple drinkers; wood shavings are used as litter. The moisture content of the poultry litter is approximately 25 %. Some dirty water is produced during cleanout between cycles, and this is removed and spread onto land, usually within one to two kilometres. The volume of waste water varies greatly, but is usually in the order of 1,000 litres per 15,000 – 20,000 birds.

All poultry litter from the the six poultry facilities was sold to a contractor up to a date prior to Year 3 and Year 4, with more than 80 % being used locally (but not in the immediate vicinity of the farm), with the rest going further afield. At Grampian, no poultry manure was applied within 1 km distance from the farm to avoid animal health risks. After that date, more than 95 % of all litter went to be combusted for electricity generation.

7.3.3 Derivation of emission potentials for the case study

In the AENEID model, the NH_3 emission estimates are based on emission potentials derived from Misselbrook *et al.* (2004). These emission potentials are based on average farming practice, without taking into account regional variations that may occur. Management practice such as drinker systems and type of bedding material may affect the litter moisture content, and therefore the subsequent NH_3 emission potential. The detailed information provided for each poultry farm in the case study made it possible to reduce uncertainties associated with the emission potentials applied in the study. The poultry emission potentials from Misselbrook *et al.* (2004) shown in Table 7.1 were therefore modified with management information provided by Grampian.

Table 7.1. Emission potentials (EP) for broilers and layers derived from Misselbrook *et al.* (2004).

	Broilers (indoor) **EP** (kg N 1000 birds^{-1})	**Layers (indoor)** **EP** (kg N 1000 birds^{-1})
Grazing	0	0
Housing	71	190
Storage	1.4	2.9
Spreading	64.3	181
Total emission potential	136.7	374.1

Broilers, free-range broilers, broiler breeders and perchery layers were present in the six poultry facilities during the study period (1985 to 2005). The assumptions used to derive the emission potentials for these poultry types are summarised below.

Emission potentials for broilers

Nicholson *et al.* (2004) showed that the emission potential of broilers on wood shavings was 50 % smaller compared with broilers on straw during winter (but the same during summer), and that nipple drinkers are associated with lower emission potentials than bell drinkers. All broilers in the case study were kept on wood shavings and using nipple drinkers, compared with IAEUK where all broilers were assumed to be kept on straw. It is therefore suggested that the housing emission potential of broilers in the case study should be 75 % of the emission potential applied in the IAEUK. (A reduction of 50 % over a 6 month winter period is equivalent to an annual reduction of 25 %.)

When Nicholson *et al.* (2004) compared storage emissions and manure application emissions from different management systems (litter types and quantities and design of broiler drinkers), they did not find any differences. The emission potentials for storage and manure application for broilers used in Misselbrook *et al.* (2004) were therefore applied in the case study.

Emission potentials for free range broilers

Although Misselbrook *et al.* (2004) assumes that all broilers are kept indoors, the emission potential for manure dropped outdoors has been estimated in the inventory at 0.125 kg N bird^{-1}yr^{-1}. Misselbrook *et al.* (2004) assume that 12 % of excreta are dropped outdoors by free-range poultry. In the case study, the outdoor emission potential was therefore assumed to be 12 % of 0.125 kg N bird^{-1} yr^{-1} (0.015 kg N bird^{-1} yr^{-1}) and the emission potentials of housing, storage and spreading were assumed to be 88 % of the emission potentials of indoor broilers.

Emission potentials for broiler breeders

The emission potential of broiler breeders is higher than for broilers, due to a higher manure output, and the manure output of broiler breeders is therefore more similar to that of layers (M. Stevens, Grampian Country Food Group, pers. comm., 2004). The emission potential for broiler breeders in this study was therefore estimated as the average emission potential of indoor broilers and perchery layers (Table 7.2).

Table 7.2. Summary of the adjusted emission potentials (EP) applied in the case study.

	Broilers (indoor) EP (kg N 1000 birds^{-1})	Broilers (outdoor) EP (kg N 1000 birds^{-1})	Broiler breeders EP (kg N 1000 birds^{-1})	Layers (perchery) (kg N 1000 birds^{-1})
Grazing	0	15.0	0	0
Housing	53.3	46.9	136.5	219.7
Storage	1.4	1.2	2.4	3.4
Spreading[a]	64.3	56.6	122.9	181.4
Spreading[b]	14.8	13.0	28.3	41.7
Total EP (a)	119	119.7	261.8	404.5
Total EP (b)	69.9	76.1	167.2	264.8

a) Years 1 and 2 (35 % manure to powerstation)
b) Years 3 and 4 (95 % manure to powerstation)

Emission potentials for layers

All layers in the case study are perchery layers. The emission potential applied was therefore assumed to be the same as for layers kept in percheries in IAEUK (2004).

Emission potentials for spreading emissions

After Years 3 and 4 of the study period, more than 95 % of the poultry litter on the poultry facilities were exported to a power station in Fife (Scotland). Incinerating poultry manure is associated with small emissions of ammonia. The spreading emission potential for poultry litter after this date should therefore be reduced by 77 % (to compensate for the 35 % emission reduction due to incineration already made in the IAEUK). For example, if the original emission from landspreading is 100 kg N ha^{-1} yr^{-1}, a reduction by 95 % to 5 kg N ha^{-1} yr^{-1} is equivalent to a 77 % reduction of the IAEUK equivalent of 65 kg N ha^{-1} yr^{-1} as a reduction by 50 kg N ha^{-1} yr^{-1} (77 % of 65) equals 5 kg N ha^{-1} yr^{-1}.

7.3.4 Comparison of poultry populations in the study area with the agricultural census data

The case study made it possible to assess uncertainties associated with the current method of using the agricultural census data to provide information on the location of farm building emissions in AENEID. In Scotland, the agricultural census data are aggregated at parish level. Actual data on animal numbers at the case study farms were compared with those available from the census, and the errors in spatially locating housing and storage emissions within parishes at the 1-km resolution of the present model were assessed.

When comparing the two datasets, it was evident that the numbers of birds in the study area were different from the poultry numbers reported in the census (see Table 7.3). In parishes where census numbers are greater than case study numbers, it may be that poultry farms other than the study farms may be located in the parish. It is however more difficult to explain why fewer birds are reported in the agricultural census than in the case study data. This suggests that, although the actual poultry buildings are located in that parish, the poultry may be reported elsewhere, possibly where a main office is located. It may also be the case that some parishes have 'acquired' poultry from other parishes due to main office reporting. Altogether, 3,069,594 broilers were reported in the detailed case study data for Year 3, but in the corresponding parishes in the agricultural census data, only 2,389,048 broilers were reported, i.e. 680,546 (22 %) fewer birds than in the detailed data (See Table 7.3). This suggests that the agricultural census is likely to be associated with large statistical uncertainties, as well as uncertainties in the location of poultry farms.

Table 7.3. Comparison of broiler numbers in the detailed case study with the corresponding parishes in the agricultural census data for Year 3.

Parish	Case study data (broilers)	Agricultural census data (broilers)	Absolute difference (%)	Ratio of census data to case study data as a %
A	1,106,503	716,592	389,911	65 %
B	692,569	708,050	-15,481	102 %
C	969,988	665,718	304,270	69 %
D	96,957	78,941	18,016	81 %
E	193,477	219,694	26,217	114 %
F	10,100	53	10,047	1 %
A – F	3,069,594	2,389,048	680,546	78 %

Uncertainties in the distribution approach of housing and storage emissions of the AENEID model were investigated by distributing NH_3 emissions from the six poultry facilities (located in 14 1 x 1 km grid cells) in the case study. The assumption in the AENEID model, that poultry housing and storage emissions were located on suburban landcover within the parish, was tested by locating the 1 x 1 km grid cells where the poultry buildings in the case study resided. Of these 14 grid cells, 9 where on suburban landcover, 4 were immediately adjacent to suburban landcover and only one was distant from suburban landcover, but not all suburban land cover contain poultry. However, this result suggests that the assumptions made in the AENEID model regarding the spatial location of poultry farms is generally a good approximation of reality.

7.4 Development of a new module for poultry in AENEID

The information provided for the detailed case study suggested that spreading all NH_3 poultry manure in the parish of origin (as assumed in the original AENEID model), does not reflect current agricultural practice. These large uncertainties were acknowledged by Dragosits (1999), who suggested two possible solutions for dealing with the spatial location of NH_3 emissions from manure spreading.

a) *A spatial decay function* – This would be the simplest and most straightforward approach. Manure spreading emissions would be applied in a circle around the estimated location of the farm, with lower application rates the further away from the centre of the distribution.

b) *An iterative process* – As with the spatial decay function, the farm would be the central point, but emissions from manure spreading would only be distributed onto suitable land in the circle that has not yet been "filled" to its capacity. The circle would therefore gradually expand until all manure has been applied. This approach requires knowledge of manure spreading from other livestock per grid cell, as well as the manure carrying capacity for each landcover type.

The second approach (b) was chosen here, as it represents reality more closely than the first. An iterative module to distribute emissions from poultry manure was therefore developed and incorporated into the AENEID model.

7.4.1 A new approach to distribute manure spreading emissions from poultry

In the standard AENEID model, housing and storage emissions are assumed to occur at the estimated location of the poultry building, i.e. within the suburban landcover of the aggregation zone. In the new approach, ammonia emissions from the landspreading of poultry manure were also distributed according to the original AENEID model for those parishes that were considered to have sufficient agricultural land, but the new iterative process was applied for those parishes with insufficient agricultural land, so that the poultry emission was distributed over a greater area.

The starting point of the new poultry manure module was to establish the "manure carrying capacity" of the aggregation zone. The total manure produced, expressed as kg N, was calculated for each aggregation zone and then divided by the areal extent of land suitable for manure spreading, i.e. arable land and improved grassland. This average application rate (in kg N ha^{-1}) was then compared with a saturation criterion of 50 kg N ha^{-1} yr^{-1}. This saturation criterion was based on the most likely application rate of poultry manure (for further details see Section 7.4.1). The saturation criterion assumes that manure (expressed as an amount of nitrogen) can only be applied up to a particular application rate (the maximum N-rate). If the calculated average application rate within the aggregation zone is lower than the saturation criterion, then the carrying capacity of that parish is sufficient, and the manure is distributed according to the original methodology, i.e. in the aggregation zone of origin. If the application rate exceeds the saturation criterion, i.e. if there is insufficient agricultural land within the aggregation zone, the manure spreading emissions are distributed by applying a new iterative process described below.

The manure output from other livestock such as cattle and pigs in the aggregation zone was also accounted for. Therefore the new methodology is dependent on relating manure from different livestock types to amounts of nitrogen. The nitrogen excretion (from manure) was calculated for each cattle, pig and poultry category, applying data on nitrogen output for each livestock category. Information on manure output was derived from the Misselbrook *et al.* (2004), and nitrogen content of the manure was derived from MAFF (2000) for cattle and pigs and from the IAEUK for poultry. The nitrogen output for poultry was calculated by multiplying the manure output per bird by the N content of the excreta, as shown in Table 7.4. The calculated N excretion for cattle, pigs and poultry is shown in Table 7.5.

Table 7.4. Calculation of estimated N excretion per bird. Manure output (fresh mass) per bird and nitrogen content per tonne of excreta were derived from IAEUK (2000).

Livestock category	NARSES category	Manure output (t N $bird^{-1}$ y^{-1})	N content of excreta (kg N t^{-1})	N excreted (kg N $bird^{-1}y^{-1}$)
Layers	n23	0.0410	16	0.656
Breeding birds	n24	0.0410	16	0.656
Broilers	n25	0.0165	30	0.495
Pullets	n26	0.0078	30	0.234
Turkeys & other poultry	n27-n28	0.0340	30	1.020

Table 7.5. Nitrogen excreted by each livestock category (kg N livestock^{-1} yr^{-1}) during housing, based on the IAEUK (2000) and MAFF (2000).

	Category	NARSES category	N excreted (kg N livestock^{-1} yr^{-1})
Cattle	Dairy cows	n1	48
	Dairy heifers in calf	n2-n3	23.6
	Beef cows & heifers	n4	34.7
	Beef heifers in calf & bulls & other cattle (1-2 yrs)	n5, n6, n8, n10	27.5
	Bulls and other cattle > 2yrs	n7, n9	34
	Calves	n11	15.9
Pigs	Sows and gilts	n14-n16	29.7
	Boars	n17	37.4
	Other pigs, >110 kg	n18	16.8
	Other pigs, 20 - 110 kg	n19-n21	12.2
	Other pigs, < 20 kg	n22	3.3
Poultry	Layers & breeding birds	n23, n24	0.656
	Broilers	n25	0.495
	Pullets	n26	0.234
	Turkeys and other poultry	n27, n28	1.020

The level of saturation for each 1 x 1 km grid cell is calculated in a saturation grid, which keeps track of the fraction of each grid cell that has been saturated with manure (expressed as nitrogen). Manure nitrogen is only applied to those grid cells where the saturation value is less than 1. For example, if the value of the saturation grid cell is 0.25, 75 % of the suitable land within that grid cell is still available for N application up to the maximum application rate (in this case 50 kg N ha^{-1}), which equates to 37.5 kg N ha^{-1} yr^{-1} of poultry manure which can be applied to that grid cell.

For cattle, the original AENEID re-distribution methodology was applied assuming that all manure is spread within the parish, but in order to incorporate manure spreading from cattle in the saturation grid, the saturation proportion of nitrogen from cattle was calculated and tracked through the model. For pigs and poultry, those aggregation zones (parishes) where saturation is not reached, are also treated according to the original AENEID methodology, and their nitrogen contribution is added to the saturation grid, together with the cattle emissions.

Parishes that have insufficient agricultural land for the spreading of pig and poultry manure are treated according to the new iterative process. In these parishes, the poultry manure N was distributed onto suitable landcover (arable or grassland), up to the maximum application rate (50 kg ha^{-1}), in a zone around the estimated location of the farm. Due to the data format (grid cells), the "zone" was "diamond shaped" rather than circular. The saturation zone was expanded in an iterative process until all nitrogen had been applied, taking into account the saturation proportion from previous spreading of nitrogen from cattle, pigs and poultry. No poultry manure was allocated to those 1-km grid cells where the poultry farm was assumed to be located (as poultry manure is normally not spread too close to the farm to avoid animal health risks, see Section 7.3.2). When all nitrogen from poultry has been distributed, the nitrogen is converted back into equivalent numbers of poultry in order to make the re-distribution result uniform with the original AENEID model.

Although the module is based on detailed information from poultry farms only, the approach was developed to include emissions from intensive pig farms, as their excess manure may also be transported beyond the parish of origin. The only difference in the pig model was the saturation threshold rate of 100 kg N ha^{-1} instead of 50 kg N ha^{-1}. The saturation rate for pig manure was set at a higher rate to ensure that the estimated transportation distance of pig manure in the model is shorter than for poultry, as pig manure is more difficult to transport long distances (see Section 7.5.1 for further details). The pig and poultry module was incorporated into the AENEID model, with pig manure being distributed before poultry manure.

7.4.2 Estimating ammonia emission maps

A spatially distributed NH_3 emission map for the six poultry facilities in the case study was calculated based on the detailed information provided (Hellsten *et al.*, 2005). The additional information made it possible to adjust the emission potentials to the actual management practices applied, therefore reducing uncertainties associated with the magnitude of emissions (Section 7.3.3). Spatial uncertainties were also reduced, as it was possible to locate housing and storage emissions at their exact location rather than at a location estimated by AENEID within the parish of origin. The spreading emissions from poultry litter were distributed by applying the new module (Section 7.4.1). Ammonia emission estimates based on the new AENEID approach were calculated and compared with the original AENEID approach where all landspreading emissions of poultry manure were distributed within the parish of origin.

7.5 Results and discussion

The poultry facilities included in the case study were considered to give a representative approximation of intensive poultry farms in Scotland as they are located in three out of the top four poultry parishes in Scotland. These poultry farms are also considered to be representative for the main part of the poultry industry in the UK of similar poultry farming size, management practice and proximity to poultry incineration stations. However, additional case studies on poultry farms in other parts of the UK would further reduce uncertainties in the modelling result.

Due to confidentiality reasons, the estimated spatial distribution of ammonia emissions from the six poultry facilities (Hellsten *et al.*, 2005) cannot be shown in a geographical context where the exact location of the farms may be revealed. Figure 7.1 shows the effect of applying different distribution approaches on two poultry facilities located in neighbouring parishes in the case study area. Figure 7.1.a shows the 1 x 1 km grid cells where the two poultry facilities are located. In Figure 7.1.b, the original AENEID distribution methodology has been applied, and all emissions are located in the parish of origin. Figure 7.1.c. shows the distribution using the new iterative methodology for manure spreading emissions.

The new AENEID approach to distribute emissions from poultry is based on a combination of the original approach (Figure 7.1.b) and the new distribution methodology (Figure 7.1.c). The new methodology is only applied to those parishes that have an insufficient amount of agricultural land for spreading of the poultry manure produced in the parish, while parishes with sufficient agricultural land are treated according to the original approach. The result of applying the new AENEID approach is shown in Figure 7.1.d.

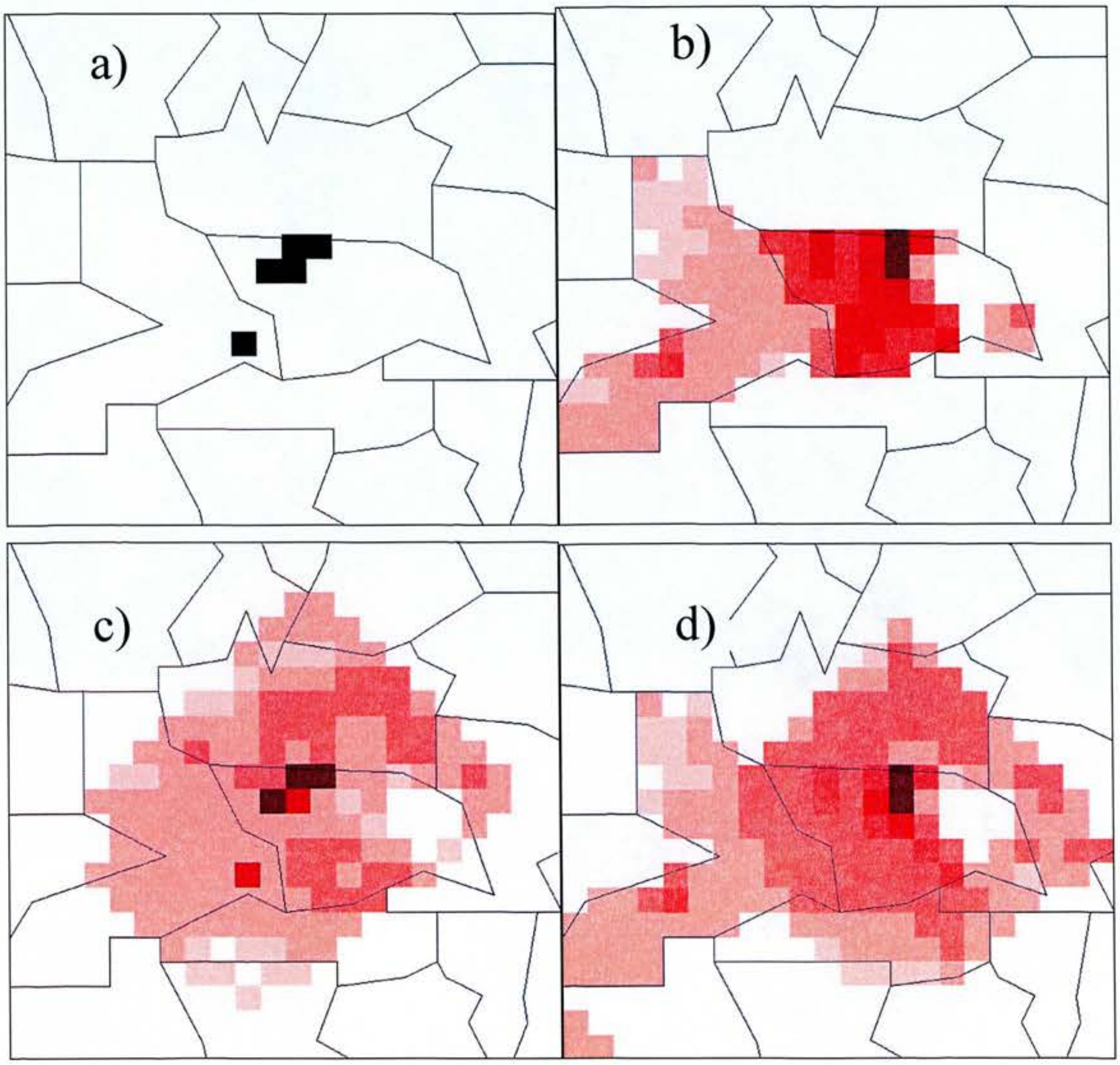

Figure 7.1. Landspreading emissions from two of the poultry farms included in the detailed study, resulting from applying different distribution methodologies. a) The 1 x 1 km grid cells where the two poultry facilities are located. (The poultry buildings of the two farms are located in five grid cells.) b) Applying the original AENEID distribution approach with the farm locations estimated by the model. c) Applying the new iterative distribution process at the exact location of the farms. d) Applying the new AENEID approach with the farm locations estimated by the model and the new distribution process applied to those parishes with insufficient agricultural land. (Note, some locations and coordinates were changed to maintain confidentiality).

In Figure 7.1.d, the emission pattern from the farm on the right is diamond shaped, i.e. the new distribution approach has been applied, whereas the spreading pattern of the farm on the left (in a different parish) is similar to the emission pattern in Figure 7.1.b, i.e. the emissions were distributed using the original approach, as sufficient agricultural land within the parish was available.

In grid cells within the diamond-shaped spreading zones that show zero emission (Figure 7.1.c and Figure 7.1.d), no poultry manure was applied, either due to lack of suitable agricultural land, or because the grid cells were already saturated with manure from other livestock such as cattle and pigs.

When comparing Figure 7.1.c and Figure 7.1.d, it can be seen that the spreading pattern of the diamond shape is different. This is because in Figure 7.1.d, the location of the poultry farms within each parish applied in the AENEID model is estimated by the model, rather than at the exact location of the case study farms as in Figure 7.1.c. The approximate locations of farms in Figure 7.1.d. are then used as the centre point for the diamond-shaped spreading zone, while the centre point in Figure 7.1.c is situated at the exact location of the case study farms. When comparing the four emission maps in Figure 7.1 it can be seen that the new module smoothes out the emission peaks, compared with the original AENEID approach. The best estimates are provided by Figure 7.1.c, although this method may only be applied where knowledge of the exact farm location is available. Table 7.6 shows the estimated ammonia emission from poultry in the case study for the years Y1, Y2, Y3 and Y4. Differences are due to different numbers and types of birds during the study period, but also due to differences in the emission potentials applied (Table 7.1 and Table 7.2). The emission estimates for Year 3 change significantly depending on type of emission potentials applied. The emission estimate of 419.6 kt NH_3-N yr^{-1} based on the average IAEUK emission potentials (Table 7.6) is the largest, whereas taking account of local management practices reduced emissions significantly (Table 7.2). Overall, the most important factor affecting the emission potential for broilers in the study area was the amount of manure that was incinerated.

Table 7.6. Total ammonia emission from poultry included in the case study, applying the adjusted emission potentials from Table 7.2.

Year	Total emission from poultry NH_3-N (kg yr^{-1})
Year 1	457.0
Year 2	446.2
Year 3 (a)	366.7
Year 3 (b)	254.5
Year 3 (c)	419.6
Year 4	199.3

a) Assuming 35 % of the poultry manure incinerated
b) Assuming 95 % of the poultry manure incinerated
c) application of emission potentials of the original approach (Table 7.1).

Ammonia emissions from other livestock (cattle, sheep, pigs etc.) were also calculated and added to the 1 x 1 km map of poultry manure, based on both the original and the new AENEID approach. The final emission maps were aggregated to a coarser resolution (5 x 5 km) for visualisation, and to reduce some of the uncertainties in the modelling process. The 1-km map based on the original AENEID approach (Figure 7.2.a) clearly shows those parishes with poultry farms as hotspots, and these hotspots are still visible in the 5 x 5 km map (Figure 7.2.b). When the livestock emission map was based on the new AENEID approach where the new module was incorporated (Figure 7.3), the emission pattern is much smoother, with significantly fewer emission peaks as an artefact of the original approach for poultry locations. Figure 7.4 shows the best possible emission map, where the detailed information has been fully incorporated, both regarding the exact location of poultry farms, and regarding the emission potentials that have been updated. Due to confidentiality constraints, the 1-km resolution emission map for Figure 7.4 cannot be shown here, as this would reveal the location of the poultry farms. When comparing the emission result of the new AENEID model (Figure 7.3.b) with the emission result based on the case study information (Figure 7.4), it can be seen that the new iterative AENEID approach comes much closer to the case study situation than the original AENEID approach (7.2.b).

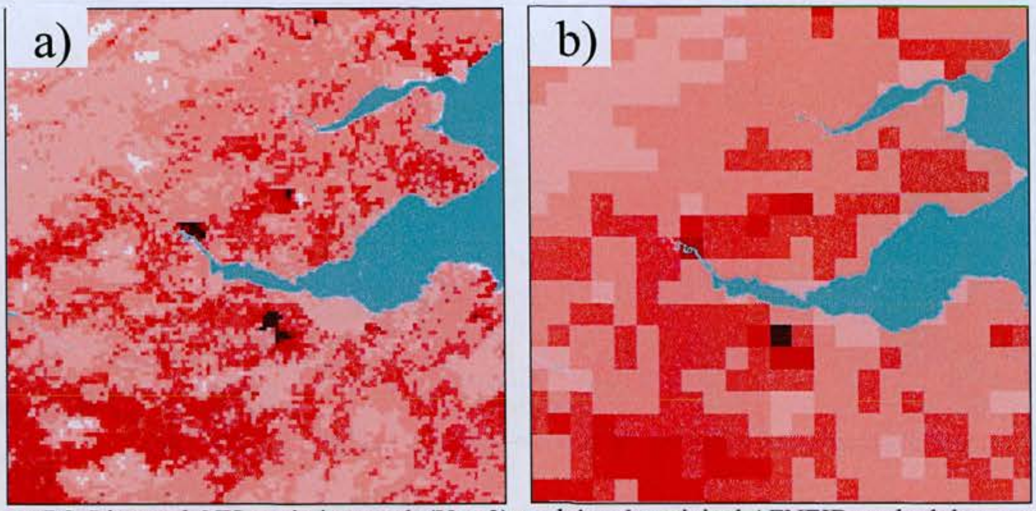

Figure 7.2. Livestock NH_3 emission result (Year3) applying the original AENEID methodology at a) 1-km resolution, and b) 5-km resolution

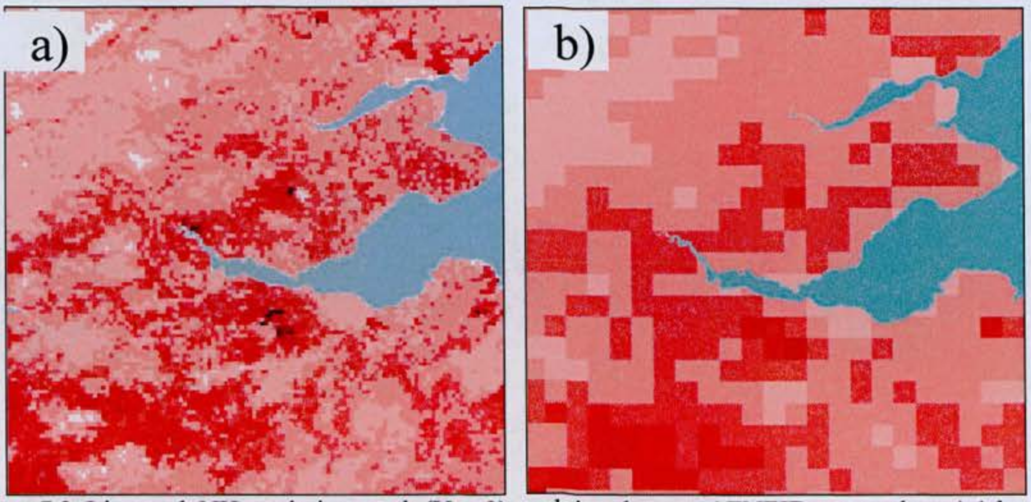

Figure 7.3. Livestock NH_3 emission result (Year3) applying the new AENEID approach at a) 1-km resolution, and b) 5-km resolution.

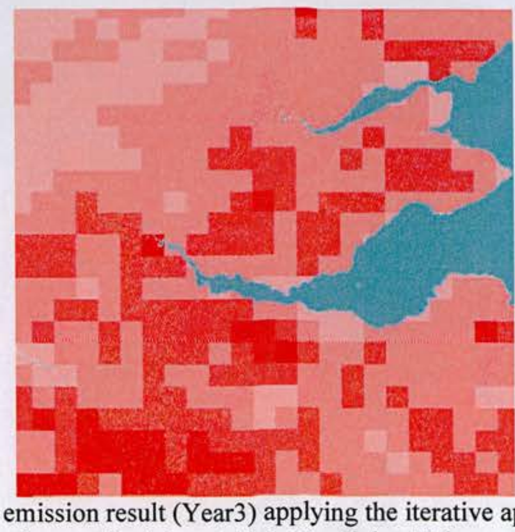

Figure 7.4, Livestock NH_3 emission result (Year3) applying the iterative approach at the exact location of poultry farms and new adjusted emission potentials at 5-km resolution. The 1-km map can not be shown due to confidentiality issues regarding the exact location of the farms.

Table 7.7 shows the maximum livestock emission in any gridsquare in Scotland for Year 3, comparing the original AENEID approach with the new AENEID distribution methodology. At the 1 x 1 km resolution, the new approach results in lower emission peaks at both the 1-km and 5-km resolution, as a consequence of the iterative process that distributes the landspreading emissions over a wider area.

Table 7.7. The maximum livestock emission for Year 3 for both the original and the new AENEID approach. The emission potentials are based on Table 7.1.

Approach	Maximum emission from livestock (1-km grid resolution) NH_3-N (kg $ha^{-1}yr^{-1}$)	Maximum emission from livestock (5-km grid resolution) NH_3-N (kg $ha^{-1}yr^{-1}$)
Original AENEID	17.8	8.6
New AENEID	14.8	6.5

7.5.1 Sensitivity to the saturation rate criterion

The largest uncertainty regarding the new distribution methodology is the saturation rate, i.e. the maximum application rate of manure nitrogen applied to agricultural fields. According to current fertilizer recommendations (MAFF, 2000), manure nitrogen applied to crops and grassland should not exceed 250 kg N $ha^{-1}yr^{-1}$. It should be borne in mind that the saturation calculation in the model only includes manure application from cattle, pigs and poultry, while other nitrogen sources, such as grazing animals, application of mineral fertilizers and application of manure from other livestock types (sheep, horses and goats) were not included in the saturation grid. In this study, it is therefore suggested that the saturation level should be set at a much lower level than 250 kg N $ha^{-1}yr^{-1}$, to allow for the potential contribution by other nitrogen sources. Furthermore, not all farmland located close to a poultry farm will acquire poultry manure from the farm, and in most cases poultry manure would not be applied annually. Typically, (B. Chambers, ADAS, pers. comm., 2005), manure is applied to those fields that do receive poultry litter at a rate of ~200 kg N ha^{-1} yr^{-1}, and these fields typically only receive it e.g. every second year. If it is then assumed that only 50 % of the fields close to the poultry farm receive poultry manure, this results in an average effective saturation rate of 50 kg N $ha^{-1}yr^{-1}$. Furthermore, the assumption averages out where some land receives large amounts of poultry manure (up to 250 kg N ha^{-1} yr^{-1}), while other land may not receive any poultry manure at all, despite being situated close to a poultry farm. The saturation rate is therefore not strictly speaking representing reality, but rather the average rate. Manure from cattle and pigs was assumed to be applied up to a higher maximum threshold of 100 kg N ha^{-1} yr^{-1}, which is deemed as a suitable application rate for these types of manure (B. Chambers, ADAS, pers. comm., 2005). Furthermore, as pig manure is more difficult to transport long distances than poultry manure due to the greater volume, and setting a higher saturation rate ensures that the estimated transportation distance of pig manure in the model is shorter.

In order to get an indication of the suitability of a maximum application rate of 50 kg N ha^{-1} yr^{-1}, a sensitivity analysis of applying different maximum saturation rates for poultry manure was carried out for the poultry farms in the case study area. Emission maps for poultry were calculated based on a saturation rate of 100 kg N ha^{-1} yr^{-1} for cattle and pigs, while different saturation rates of 25 kg N ha^{-1} yr^{-1}, 50 kg N ha^{-1} yr^{-1} and 100 kg N ha^{-1} yr^{-1} were applied for poultry. From Figure 7.5 it can be seen that lower saturation rates require more suitable agricultural land, therefore increasing the areal extent over which the model distributes the poultry litter. The transportation distance of the poultry manure was compared with information on the likely distance of manure transport provided in the poultry case study, where > 80 % of the manure is being used locally but not adjacent, with the rest going further afield. The furthest distance the poultry litter was transported was approximately 50 km (B. Edwards, Richard Edwards & Sons, pers. comm., 2004). Based on this information, a maximum saturation rate of 50 kg N $ha^{-1}yr^{-1}$ was considered as providing the most realistic transport distances for poultry manure, and was therefore adopted as the saturation criterion for poultry manure in the module.

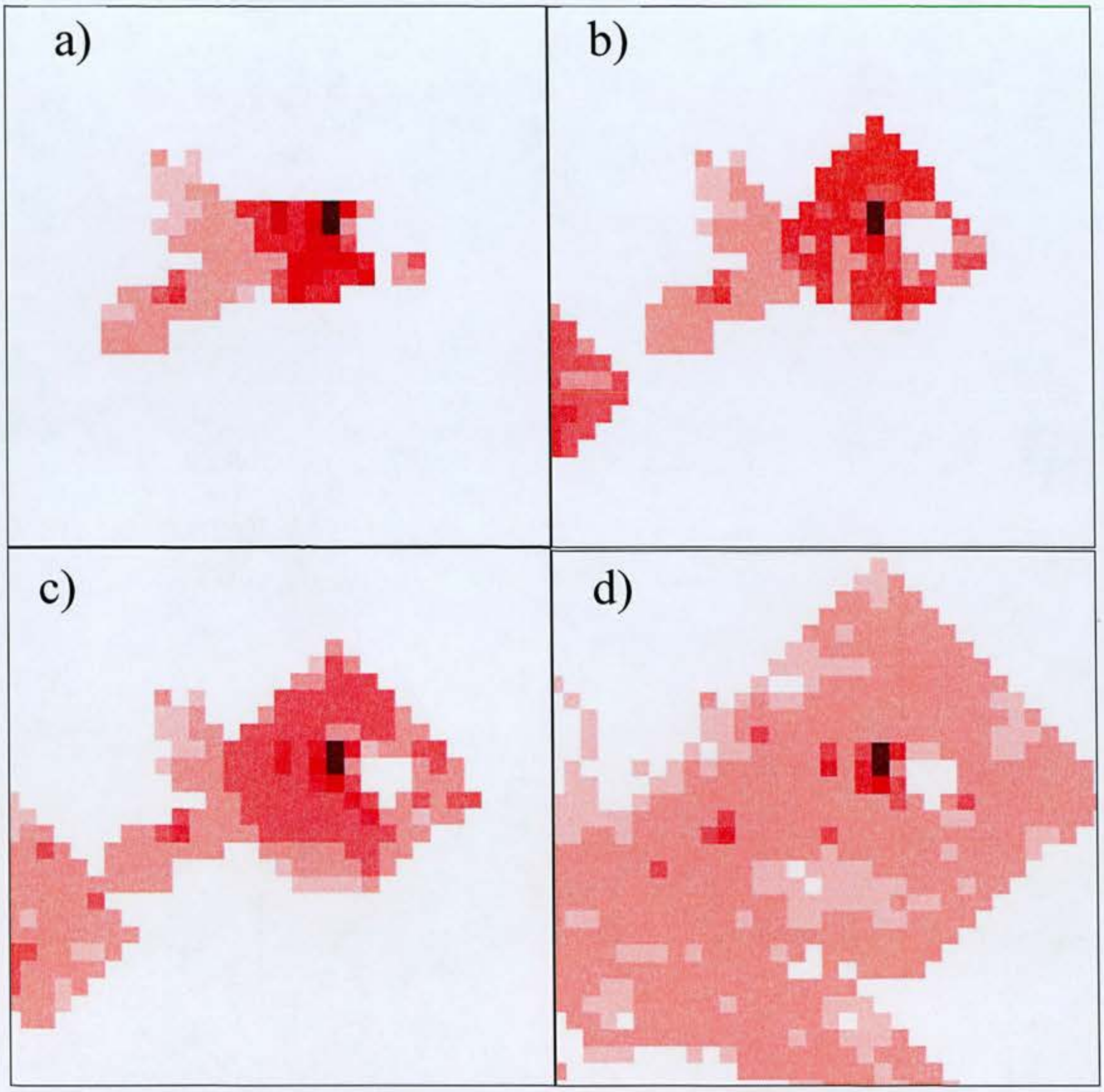

Figure 7.5. The effect of applying various maximum application rates (1-km grid resolution). a) Original AENEID methodology. b) Poultry saturation criterion: 100 kg N ha^{-1}. c) Poultry saturation criterion: 50 kg N ha^{-1}. d) Poultry saturation criterion: 25 kg N ha^{-1}.

The saturation rates of 50 kg N ha^{-1} yr^{-1} for poultry and 100 kg N ha^{-1} yr^{-1} for cattle and pigs were further investigated by calculating average application rates per ha of agricultural land in the aggregation zones for cattle, pigs and poultry. Figure 7.6 shows the result of this calculation based on the original AENEID approach. Red areas contain insufficient suitable agricultural land within the aggregation zone, i.e. the amount of nitrogen from cattle, pigs and/or poultry exceeds the saturation rate of 100 kg N ha^{-1} yr^{-1}. More red areas (i.e. areas exceeding the saturation rate) were anticipated, but from Figure 7.6. it can be seen that, with a lower saturation rate of 50 kg N ha^{-1} yr^{-1}, a significantly greater area would exceed the saturation rate (dark blue areas). Figure 7.6 suggests that many areas in the UK exceed the maximum manure N application rate when the original AENEID approach is applied.

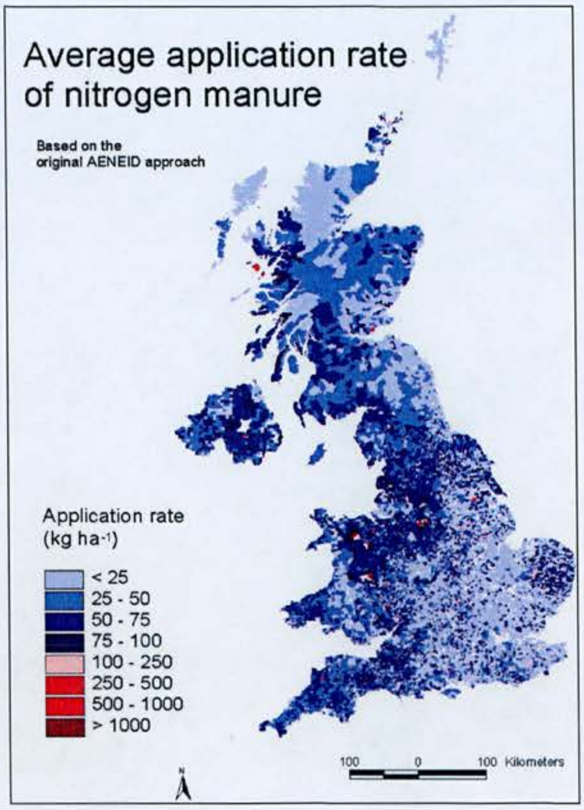

Figure 7.6. Average nitrogen application rate (from cattle, pigs and poultry) for agricultural land in the aggregation zones when the original AENEID approach is applied.

In Figure 7.7.a, the average application rate in the aggregation zones is based on the new module for pig and poultry manure. It can be seen that the number of aggregation zones exceeding the saturation rate is smaller than in Figure 7.6: red areas in Figure 7.7.a are therefore dominated by cattle nitrogen, as the new approach smoothes out the emissions from pigs and poultry. The red areas in Figure 7.7.a may therefore indicate that these aggregation zones are insufficient for the landspreading of cattle manure. It is likely that these parishes still have enough agricultural land available, e.g. semi-improved areas may take some of the manure. It could also be dependent on the aggregation of the agricultural census data, i.e. due to the Modifiable Areal Unit Problem (MAUP) explained in Chapter 9. The average application rate was therefore calculated, where manure nitrogen was distributed onto all agricultural land within the aggregation zone (Figure 7.7.b) rather than just arable land and intensive grassland. In this scenario, no areas within the UK exceeded the saturation rate, suggesting that aggregation zones with insufficient suitable agricultural land still have enough agricultural land to assimilate the cattle manure. The problem could also be due to some uncertainty in the LCM 2000 grassland categories, as the areas exceeding the threshold in Figure 7.7.a are mostly located in hill areas.

a) b)

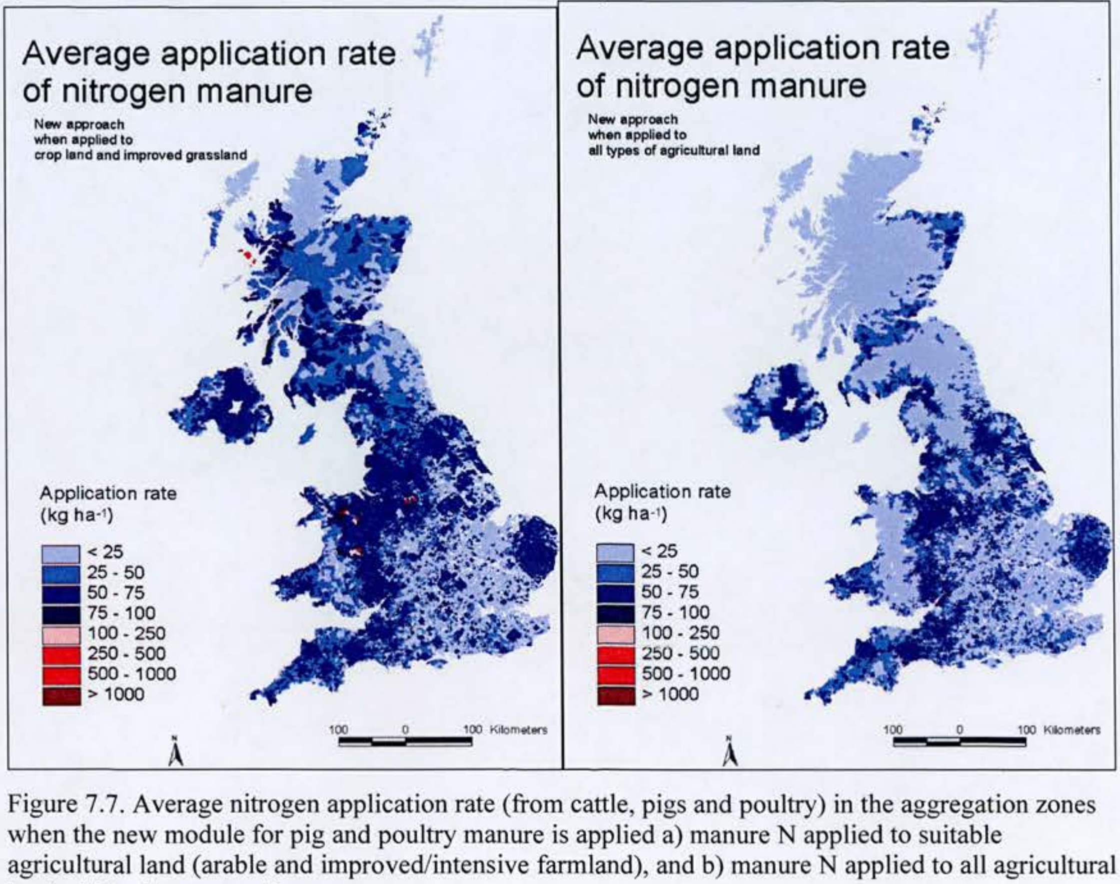

Figure 7.7. Average nitrogen application rate (from cattle, pigs and poultry) in the aggregation zones when the new module for pig and poultry manure is applied a) manure N applied to suitable agricultural land (arable and improved/intensive farmland), and b) manure N applied to all agricultural land within the aggregation zone.

7.5.2 1-km resolution emission maps

Currently the ammonia emission maps are aggregated to a 5 x 5 km grid resolution for distribution and application in atmospheric transport models. The use of NH_3 emission maps at a 1 x 1 km resolution instead would provide the basis for a more robust environmental risk assessment through their effect on atmospheric nitrogen deposition estimates. The basic principles of the AENEID approach ensure that confidentiality of the data input is maintained by linking emissions to land cover within parishes, so that the emission for each component at a 1-km level cannot be directly related to numbers of animals from any farm. The availability of suitable agricultural landcover varies within the concentric spreading zone, resulting in variable emissions of ammonia.

It may however be argued that using the results of the new module at a 1-km grid resolution while maintaining confidentiality of the underlying farm input data could prove challenging, because the diamond shape occurring as a result of the distribution approach reveal the location of large poultry farms. Confidentiality should however still be ensured, because the centre of the diamond is not where the poultry farm is located, but is instead an approximation of the location of the poultry farm by the model. Furthermore, the AENEID modelling results are only reported in aggregated form, i.e. combining different source stages (housing, storage, grazing, manure spreading) and different source types (cattle, pig, poultry, sheep, fertilizers, non-agricultural sources etc), therefore masking the diamond-shape as shown in Figure 7.8.

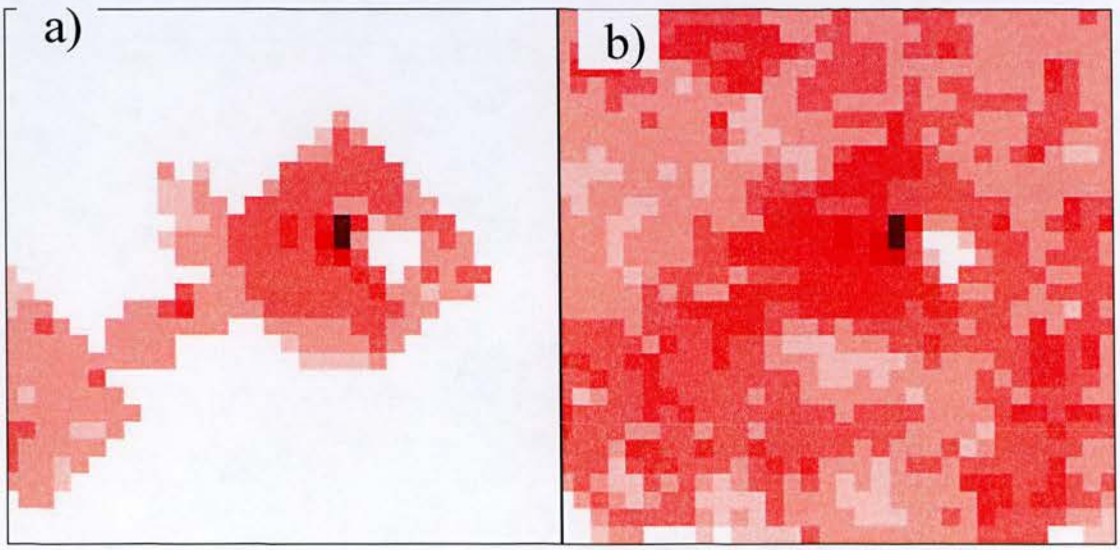

Figure 7.8. The effect of adding NH_3 emissions from other agricultural sources to the distribution of poultry emissions, hence masking the diamond shapes that may arise at 1 x 1 km level as an artefact of the new distribution approach. a) poultry emissions only b) all agricultural emissions.

When the poultry farms in the case study were modelled by applying the new approach at a 1-km grid resolution, it was not possible to detect the diamond shapes in the final 1-km grid map, as the additional emissions from other livestock and the variability in landcover within the spreading zone masked the diamond shape. However, if it is considered that these operations are not sufficient to ensure confidentiality, a decay function could be incorporated into the model to mask the diamond shape further. The decay function would smooth out the edges of the diamond, and in combination with the masking effect of the other agricultural NH_3 emission sources, confidentiality would be ensured even at the 1-km grid level. The decay function could, for instance, be based on an assumption that the saturation rate of nitrogen manure is reduced with increasing distance from the farm according to a continuous function, or according to the following:

- 50 % of the poultry nitrogen is applied at 100 % of the saturation rate closest to the farm.
- 25 % of the poultry nitrogen is applied at 75 % of the saturation rate further away from the farms.
- 15 % of the poultry nitrogen is applied at 50 % of the saturation rate even further away.
- 10 % of the poultry nitrogen is applied at 25 % of the saturation rate furthest away from the farm.

It may be argued that a decay function, where the application rate declines with distance, from the farm would also further improve the modelling result of the module. It is however questionable whether this really reflects actual farming practice.

7.5.3 Adjusting emissions for incineration of poultry manure

The detailed case study showed that the choice of emission potentials has a significant impact on the final emission result (see Table 7.6). The emission potential for poultry is influenced substantially by the amount of manure that is incinerated, as the emission potential from the combustion process is negligible compared with landspreading of poultry manure. Ammonia emissions from incineration are assumed to be zero and are therefore completely removed from the assessment. It may be argued that there are unaccounted emissions from incineration and from losses while poultry manure is in storage. Currently, however, there are no reliable data to evaluate ammonia emissions from incineration (T. Misselbrook, IGER, pers. comm., 2006).

Currently, 95 % of the manure from the poultry farms included in the case study is incinerated. This is a significantly larger manure proportion than the 35 % assumed to be incinerated on average in Misselbrook *et al.* (2004). If 95 % of the poultry manure in IAEUK were incinerated, the total poultry emissions in the UK would be reduced by 12.6 % (see Table 7.8). This suggests that it is important to include accurate estimates of the proportion of poultry manure that is incinerated, and preferably also the location of poultry litter powerstations so that spatial variations in poultry litter incineration across the UK can be accounted for. If the location of the powerstations is known, manure incineration could be modelled based on consumption of litter by powerstations and a radius around each power station.

Table 7.8. The hypothetical effect of incinerating 35 % of broiler manure on total poultry emissions in the UK, compared with the 95 % from the detailed study.

Approach	Incinerated broiler manure (%)	Poultry manure landspreading NH_3–N (kt)	Poultry emissions NH_3–N (kt)
IAEUK approach	35 %	16.5	45.4
Case study approach	95 %	11.8	39.7

7.5.4 Distinguishing between manure spreading on grass and tillage in the new module

It may be argued that the proportion of manure applied to grass and arable land, respectively, in the new distribution methodology should be distinguished. In Misselbrook *et al.* (2004) 53 % of the poultry litter is assumed to be applied to tillage and the remaining 47 % to grassland, reflecting UK average conditions. The AENEID module however, assumes that poultry manure is applied to the fields close to the farm regardless of whether they are grass or arable fields. Distinguishing the proportion of manure N applied to arable and grassland would result in different transport distances for manure spread onto grass and tillage, depending on the proportion of these landcover types in an area. This was tested for the UK. The largest difference in transport distances was found in East Anglia, i.e. manure applied to grassland had to be transported further than manure spread onto arable land, due to the limited amount of grassland available in the area. A distinction between grass and tillage was therefore not incorporated into the model, as it is more likely that the local conditions and type of available agricultural land influence the proportion of manure application to grass and arable. Furthermore the proportion of manure applied to grass and arable in the IAEUK is very similar (47 % compared with 53 %) on average.

7.6 Conclusions

In the original AENEID model (Dragosits, 1999), all NH_3 emissions were assumed to occur in the aggregation zone of origin. This assumption resulted in unrealistically high localized emission peaks, especially for poultry, due to the limited amount of suitable land available for manure spreading within the parish of origin. These modelled "hot spots" were regarded as artefacts due to limitations of the modelling approach.

In this chapter, a new mapping approach for landspreading of pig and poultry manure was developed as a module within AENEID, to reduce some of the spatial uncertainties associated with this emission source. The module is based on an iterative process where pig and poultry manure in aggregation zones with insufficient land suitable for manure spreading is transported further away from the farm, using manure "saturation rate". This iterative process takes into account manure spreading from other livestock types, and expands the area around the pig and poultry farms until all manure has been applied. The manure saturation criterion is based on a maximum likely rate of manure nitrogen application to the field.

The emission maps based on the new AENEID module show that extreme emission values are smoothed out, and land spreading emissions from pigs and poultry have been moved outside the aggregation zone of origin where insufficient suitable land is available for spreading. The new approach has been shown to be more realistic in a detailed case study with data from large poultry farms in Scotland, as the manure is applied at more reasonable application rates over a wider area than in the original model.

The case study of poultry farms in Scotland used detailed information for six poultry facilities and served to demonstrate that the ammonia emission estimate could be improved significantly both regarding uncertainties in the magnitude and the spatial location of emissions when detailed data are included in the modelling process. The additional information on management practice made it possible to modify the average UK emission potentials applied in the model, hence reducing uncertainties in estimating ammonia emissions from poultry. The amount of poultry manure that is incinerated was identified as the most important uncertainty when estimating the emission potential from poultry in the case study area. However, currently emissions from incineration are assumed to be zero, and future studies should therefore concentrate on assessing ammonia emissions from poultry power stations. Information on the exact location of poultry farms reduced uncertainties regarding the spatial location of ammonia emissions from poultry housing and manure storage as the original model estimated an approximated location within the parish where the census data for the farm are returned.

Furthermore, the case study made it possible to test the assumption of allocating poultry housing and storage emissions to suburban landcover in the original AENEID model, as these modelled locations were compared with the exact locations of poultry farms in the detailed study. The comparison suggested that the assumptions in AENEID model approximate the location of emissions from poultry farms rather well. The two datasets were also compared regarding poultry numbers in the corresponding parishes. This comparison however, suggests that the agricultural census is likely to be associated with significant statistical uncertainties, as well as uncertainties associated with the location of poultry farms.

Future studies should concentrate on developing methods to combine site-based information (where available) with parish-based information, and consider new methods for mapping in the future, once the IPPC regulations for large pig and poultry farms, come into operation. The IPPC regulations are likely to make more detailed information on farm locations and farm practices available. New mapping methods could therefore allocate emissions from poultry farming based on the exact location of the poultry farms rather than the approximated location within the parish. For instance, housing and storage emissions could be allocated to the location of the farm, and the spreading emissions from these farms could be allocated using the new poultry-sub model. Furthermore, information on local farming practice could be used to derive local emission potentials which would further reduce uncertainties in the emission inventory. The incorporation of more detailed, local information such as the data provided in the poultry case study, or data potentially available after the implementation of IPPC is therefore expected to improve emission estimates of NH_3 significantly.

8 Application of the new AENEID model to describe ammonia emissions for the UK

8.1 Introduction

The new AENEID model developed in this thesis was applied to calculate a spatial ammonia emission inventory for the UK for 1990, 1996 and 2000. These years were chosen because 1990 is the base year for the Gothenburg protocol and 2000 is a target year. 1996 was included in the study because this year had been modelled in the previous spatial ammonia emission estimate by Dragosits (1999), and was therefore a valuable source of information for comparing emission results of the original and new AENEID approach.

The spatial distribution of NH_3 emissions for 2000 was analysed in detail, considering the total NH_3 emission, as well as the contributions from different source types (agricultural livestock, fertilizer application and non-agricultural emissions). Furthermore, the livestock sub-sources (cattle, sheep, pigs and poultry) were also analysed. Emission results for 2000 were also compared with IAEUK (2000) to assess differences between the two ammonia emission inventories. The 1996 emission map was compared with the 1996 emission map of Dragosits (1999) to assess the potential impact of applying the new AENEID approach together with new updated datasets and information to distribute ammonia emissions.

Temporal changes in emissions and emission patterns within the 10-year study period (1990 to 2000) were investigated to evaluate the emission change. These intra-annual changes in emissions depend on changes in animal numbers, and farming practice with time.

8.2 Derivation of emission potentials for year 1990, 1996 and 2000

8.2.1 Emission potentials for livestock

Emission potentials applied in this study for the emission maps for 1990, 1996 and 2000 were based on the retrospective inventory calculations by Misselbrook *et al.* (2004), see Section 2.5.1. Hence, emission potentials from the latest IAEUK emission inventory available (year 2003) were derived and applied for all livestock categories (Table 8.1), however, modified to account for temporal changes in management activities for livestock during 1990 - 2003 (Table 8.2).

Table 8.1. Emission potentials (kg NH_3-N animal^{-1} year^{-1}) for 1990, 1996 , 2000 and 2003, and the difference (in %) compared with 1990. The categories N1 to N31 are described in Table 4.2.

	Category	1990	1996	2000	2003	Difference 1990-1996	Difference 1990-2000	Difference 1990-2003
Cattle	N1	24.14	24.57	24.41	23.90	1.8 %	1.1 %	-1.0 %
	N2, N3	13.36	13.68	13.57	13.20	2.4 %	1.5 %	-1.2 %
	N4	10.03	9.91	9.72	9.63	-1.3 %	-3.1 %	-4.0 %
	N5, N6	8.26	8.14	7.96	7.86	-1.5%	-3.7%	-4.9%
	N7, N9	9.94	9.81	9.63	9.54	-1.3%	-3.1%	-4.0%
	N8, N10	8.19	8.06	7.88	7.79	-1.5%	-3.8%	-4.9%
	N11	2.96	2.91	2.83	2.78	-1.8%	-4.6%	-6.0%
Sheep	N12	0.58	0.58	0.58	0.58	0.0%	0.0%	0.0%
	N13	0.12	0.12	0.12	0.12	0.0%	0.0%	0.0%
Pigs	N14, N15, N16	6.67	6.65	6.66	6.66	-0.4%	-0.1%	-0.1%
	N17	7.16	7.21	7.23	7.23	0.6%	0.9%	0.9%
	N18	8.31	8.09	8.04	8.04	-2.7%	-3.2%	-3.2%
	N19, N20, N21	4.92	4.77	4.74	4.74	-3.0%	-3.6%	-3.6%
	N22	0.75	0.75	0.75	0.75	0.0%	0.0%	0.0%
Poultry	N23	0.44	0.42	0.41	0.41	-4.2%	-7.0%	-7.0%
	N24	0.26	0.26	0.26	0.26	0.0%	0.0%	0.0%
	N25	0.15	0.12	0.12	0.11	-16.8%	-20.5%	-27.3%
	N26	0.09	0.09	0.09	0.09	0.0%	0.0%	0.0%
	N27, N28	0.39	0.39	0.39	0.39	0.0%	0.0%	0.0%
Horses	N29	10.6	10.6	10.6	10.6	0.0%	0.0%	0.0%
Goats	N30	0.57	0.57	0.57	0.57	0.0%	0.0%	0.0%
Deer	N31	1.16	1.16	1.16	1.16	0.0%	0.0%	0.0%

Spreading emissions from cattle decreased after 1993, due to 1 % cattle slurry being incorporated (Table 8.2). Table 8.1 shows that, as a result of this, the cattle emission potential decreased for cattle categories compared with 1990. The only exception are dairy cattle (N1-N3), where emissions increased from 1990 to 1996 and 2000, due to increased N input to pasture. Cattle grazing emissions are calculated using N input to

Table 8.2. Changes in management activities for livestock during 1990 to 2003, as reported by Misselbrook *et al.* (2004). These changes were incorporated in the IAEUK calculation of 2003, to calculate the emission potentials for 1990, 1996 and 2000 respectively.

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
% cattle slurry injected	0	0	0	1	1	1	1	1	1	1	1	1	1	1
% pig slurry injected	0	2	3	5	6	8	9	11	11	11	11	11	11	11
% dry sows outdoors	20	20	20	20	22	24	26	28	28	28	28	28	28	28
% farrowing sows outdoors	20	20	20	20	23	25	28	30	30	30	30	30	30	30
% boars outdoors	20	20	20	20	22	24	26	28	28	28	28	28	28	28
% fatteners >20 kg outdoors	0	0	0	0	0	1	1	1	1	1	1	1	1	1
% fatteners <20 kg outdoors	0	0	0	0	0	1	1	1	1	8	8	8	8	8
% fatteners >20 kg fully slatted	25	25	25	25	23	20	18	15	15	15	15	15	15	15
% fatteners >20 kg part slatted	25	25	25	25	24	23	21	20	20	20	20	20	20	20
% fatteners >20 kg on straw	50	50	50	50	53	57	61	65	65	65	65	65	65	65
% fatteners >20 kg on BAT*	0	0	0	0	0	0	0	0	0	0	0	0	0	0
% fatteners <20 kg fully slatted	35	35	35	35	33	31	29	27	27	27	27	27	27	27
% fatteners <20 kg part slatted	55	55	55	55	47	39	31	23	23	23	23	23	23	23
% fatteners <20 kg on straw	10	10	10	10	20	30	40	50	50	50	50	50	50	50
% fatteners <20 kg on BAT*	0	0	0	0	0	0	0	0	0	0	0	0	0	0
% laying hens on belt-cleaned systems	0	3	5	8	10	13	15	18	20	23	25	25	25	25
Amount of poultry litter incinerated (kt)	0	0	120	120	120	335	335	335	335	130	470	560	540	690
% pig slurry stores covered	0	0	0	0	0	0	0	0	0	0	0	0	0	0

* BAT – Best Available Technology

grazed pasture (Figure 8.1), and the emission potential for grazing was therefore modified according to the following function (Misselbrook *et al.*, 2004):

$$Y = -0.51 + 0.0742\,X$$

Y is the emission potential (in g N livestock unit^{-1} day^{-1}) and X is the N fertilizer application rate to fields grazed by the cattle (in kg N input N ha^{-1} yr^{-1}).

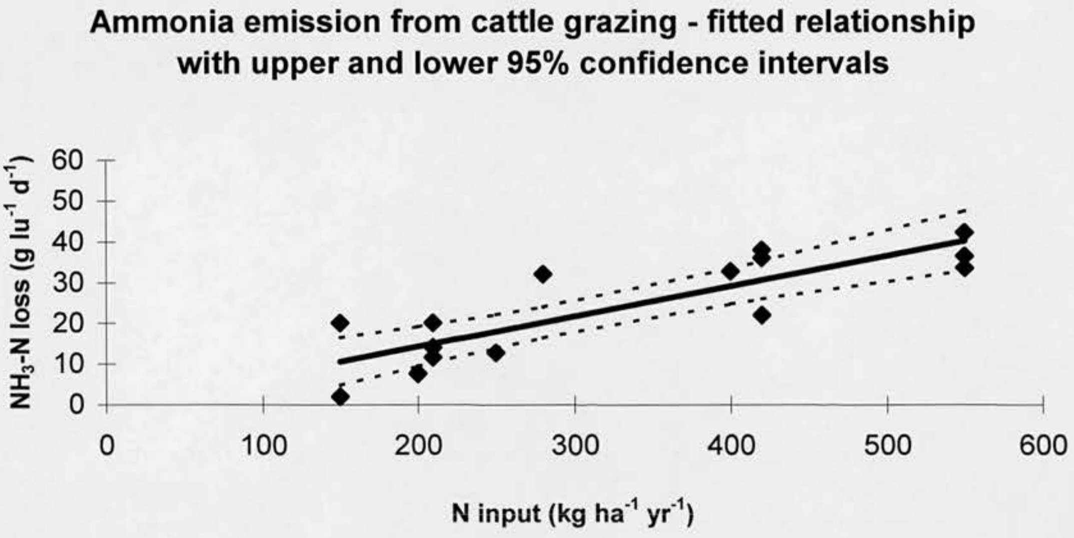

Figure 8.1 The link between N input to grazed pasture and cattle grazing emissions. Source: Misselbrook *et al.* (2004).

The emission potential for pigs has remained the same or decreased, with the exception of boars (N17) that increased due to a greater proportion of outdoor boars (Table 8.2). The increase in the number of outdoor pigs for the other pig categories was counteracted by decreases in emissions due to housing practice and increased injection of pig slurry.

The emission potential for layers decreased between 1990 and 2000 due to an increase in the proportion of layers on belt clean systems (Table 8.2). For broilers, the emission potential decreased significantly due to the increase of broiler litter being incinerated. The emission potential for all other poultry categories remained constant, as no changes in management practice were reported by Misselbrook *et al.* (2004).

8.2.2 Emission potentials for mineral fertilizers and crops

Temporal changes in emission factors from grass and crops were derived from fertilizer statistics. The N application rates to arable crops and grass respectively were calculated based on statistics for 1996 and 2000 for Great Britain (BSFP, 1997, 2001). For 1990, fertilizer statistics were not available, but were substituted with data for 1989 for England and Wales, and for 1991 for Scotland (SFP, 1990, 1992). The calculated N application rates for each crop category is summarised for all three years in Table 8.3.

Table 8.3. Fertilizer application rates (kg N ha^{-1}) for different crop and grass categories derived from SFP (1990, 1992) and BSFP (1997, 2001).

Category	Crop/Grass type	1990	1996	2000
N33	Wheat	182	154	188
N34	Winter Barley	144	94	146
N35	Spring Barley	90	185	107
N36	Oilseed rape	119	107	104
N37	Potatoes	231	188	189
N38	Other cereals	162	170	159
N39	Sugar beet	94	111	103
N40	Other root crops	73	95	71
N41	Other crops	42	32	26
N42	Vegetables	159	152	105
N43	Fruit	86	50	58
N44	Bulbs and flowers	42	32	26
N45	Grassland (< 5 yrs)	193	165	147
N46	Grassland (> 5 yrs)	98	103	87

The amounts of the different fertilizer types (ammonium nitrate, urea, UAN and others) applied as part of the overall N fertilizer during the three years are summarised in Table 8.4. The associated emission potential for each fertilizer type (expressed as a percentage of the N applied), as derived from Misselbrook *et al.* (2004), is also shown. Based on the information in Table 8.4, the overall emission potential for grassland and tillage was calculated, incorporating the different fertiliser types (Table 8.5). The emission potential from mineral fertilizers has significantly decreased since 1990, mainly as a consequence of less urea being used, but also due to generally decreased fertilizer application rates.

Table 8.4 Fertilizer use for the UK derived from BSFP (Misselbrook *et al.*, 2004). (UAN use prior to 2002 was not itemised by BSFP, so the same proportion was assumed to have been applied in previous years as in 2002.) The emission potential, expressed as a percentage of the N applied, for each fertilizer type is also presented.

	1990 (kt N yr^{-1})	1996 (kt N yr^{-1})	2000 (kt N yr^{-1})	EP (%)
Total fertilizer N applied to conserved grassland				
Ammonium nitrate	138	126	121	1.6 %
Urea	49	9	9	23 %
UAN	3.2	2.7	2.6	8 %
Other	239	224	220	1.6 %
Total fertilizer N applied to tillage				
Ammonium nitrate	508	450	446	1.6 %
Urea	144	45	66	23 %
UAN	82	68	65	8 %
Other	147	165	105	1.6 %

Table 8.5 Calculated emission potential (EP) for grassland and tillage (expressed as % of applied N).

EP	1990	1996	2000
Grassland	4.09 %	2.18 %	2.19 %
Tillage	3.81 %	2.81 %	3.17 %

8.3 Spatial distribution of ammonia emissions for 2000

8.3.1 Spatial distribution of livestock emissions

Agricultural livestock emissions in the UK are estimated by the modified AENEID approach at 194 kt NH_3-N for 2000, compared with 196 kt in IAEUK (Misselbrook *et al.*, 2004). The difference of 1.1 % between the two estimates is mainly due to small differences in livestock numbers, particularly for cattle (Table 8.6), as different statistical sources for the agricultural statistics were applied in the two studies. IAEUK is dependent on published agricultural statistics (e.g. GSS, 1997), while AENEID applies confidential statistics which are acquired directly from the devolved regions. Differences in the emission result of AENEID and IAEUK for livestock emissions were also due to small differences in the emission potentials applied, despite efforts to harmonise the emission potentials in the two approaches. For instance, despite (0.7 %) more sheep being represented in the AENEID input data, the emissions are still slightly smaller (0.2%) than for IAEUK.

Table 8.6 Comparison of ammonia emissions and livestock numbers for 2000 applied in the present study, and compared with IAEUK.

	Emission (kt NH_3-N)			Animal numbers (1000's)		
	AENEID	IAEUK	Diff	AENEID	IAEUK	Diff
Cattle	116.7	119.4	-2.3 %	10,903	11,135	-2.1 %
Sheep	14.8	14.9	-0.3 %	42,359	42,264	0.2 %
Pigs	25.5	25.4	0.5 %	6,477	6,482	-0.1 %
Poultry	33.9	33.5	1.2 %	169,927	168,973	0.6 %
Horses/goats/deer	3.1	3.1	0.6 %			
Livestock	194.1	196.28	-1.1 %			

The spatial distribution of agricultural livestock emissions for 2000 based on the new AENEID approach is shown in Figure 8.2, while the differences between the new and old model are summarised in Section 8.4. The spatial distribution was analysed further into size-categories, as shown in Table 8.7. The category with the largest percentage of grid cells (37 %) is the lowest emission category (0 - 2.5 kg NH_3-N ha^{-1} yr^{-1}). There cells are mainly located in hill and upland areas, but also in areas with low agricultural activity, i.e. in urban areas. Very high emission values (> 30 kg N ha^{-1}) occur in 1.3 % of the total number of grid squares. These grid cells are dominated by pig, poultry and cattle emissions.

Table 8.7. Analysis of NH_3 emissions from agricultural livestock for 2000: Proportion of 5 x 5 km grid squares per category.

Ammonia emission category (kg NH_3-N ha^{-1} ha^{-1})	Proportion of UK grid squares
0 – 2.5	37 %
2.5 – 5	15 %
5 – 10	21 %
10 – 20	20 %
20 – 30	6 %
30 – 40	1 %
40 – 50	0.2 %
> 50	0.1 %

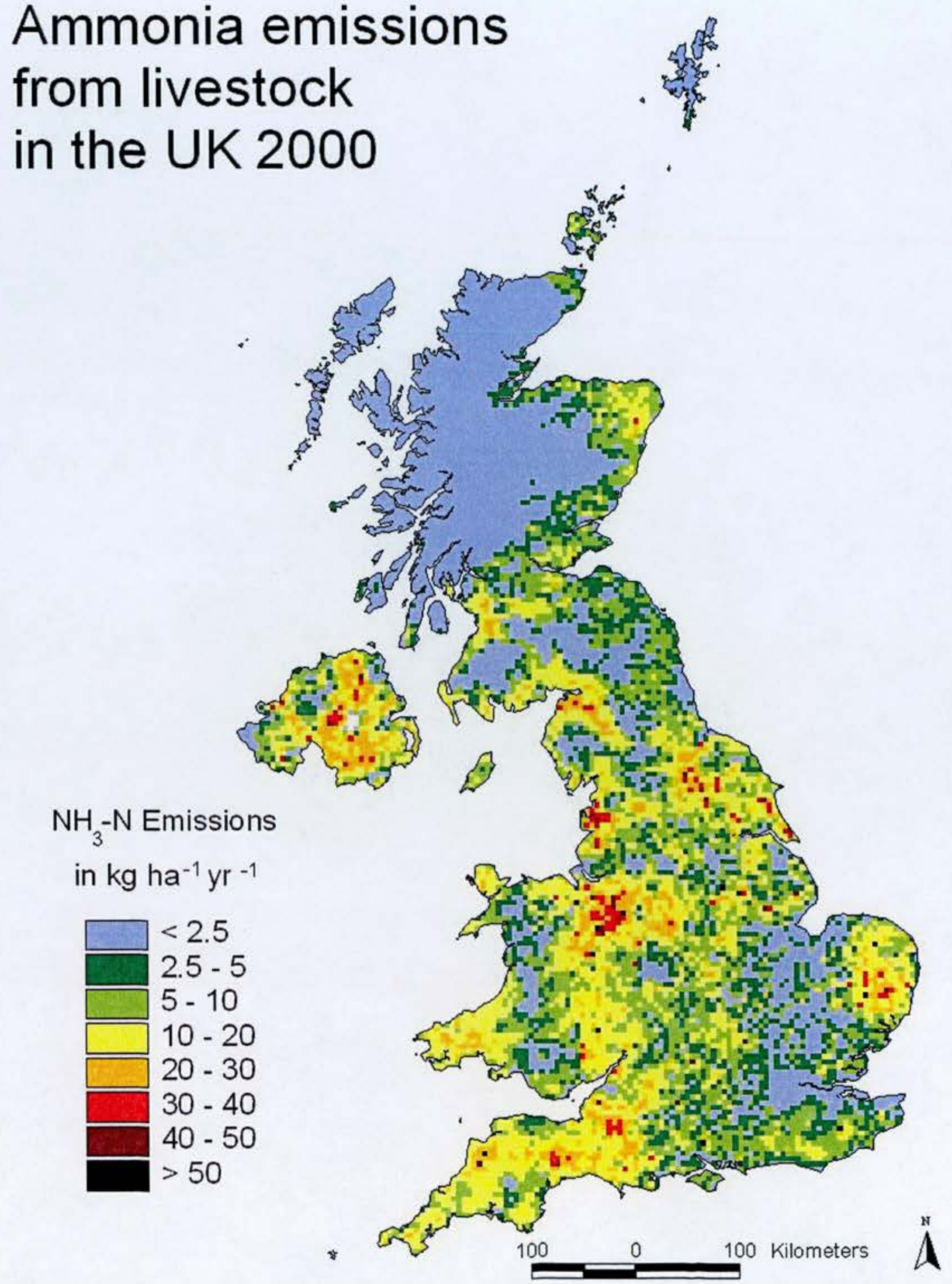

Figure 8.2. Agricultural livestock emissions in the UK for 2000, based on the new AENEID approach.

8.3.2 Spatial distribution of emissions from mineral fertiliser application and crops

Emissions from fertilizers and crops were estimated at 26.5 kt NH_3-N in the UK for 2000, which represents 12 % of the agricultural ammonia emission. Fertilizer emissions in IAEUK for 2000 were estimated at 27.1 kt NH_3-N, i.e. the AENEID estimate was 2.2 % smaller than IAEUK. IAEUK derives the N application straight from the BSFP for England, Wales and Scotland, but estimates the proportion of N on conserved grassland and tillage in Northern Ireland based on the ratio of tillage and grassland. The AENEID approach derives the N application rates to different crop categories from the BSFP (1997, 2001), and assumes that these application rates apply for the whole of UK (i.e. including Northern Ireland).

These fertiliser application rates are then applied to the different crop categories, i.e. the spatially distributed agricultural statistics. The advantage of this approach compared with IAEUK is that variations due to different types of cropping areas within the UK can be accounted for in a spatial context, which is a necessity when modelling spatial ammonia emissions from fertilizers. Finally, the tillage and grass emission potentials (from Table 8.5) were applied (using the emission potentials in IAEUK), but emissions from grass were reduced by two thirds to avoid double counting of ammonia emissions from grassland that have already been included with livestock grazing and manure spreading emissions. (For further details on the AENEID approach to calculate emissions from fertilizers, see Section 3.6). It is encouraging to find that the results of the two different approaches correspond so well (only 2.2 % difference), as the method to calculate fertilizer emissions are quite different.

The spatial distribution of fertilizer emissions in the UK (2000) is shown in Figure 8.3. Although fertilizers contribute only 12 % of the agricultural NH_3 emission, they are important in areas dominated by crop production, such as in eastern England. A large proportion of grid cells (37 %) is in the smallest emission category, and only about 0.2 % of cells exceed 5 kg NH_3-N ha^{-1} yr^{-1} (Table 8.8). Emissions from fertilized crops are located in areas dominated by arable farming in the southern and eastern part of the UK, while emissions from fertilized grass dominate in the grassland areas in the western part of the UK.

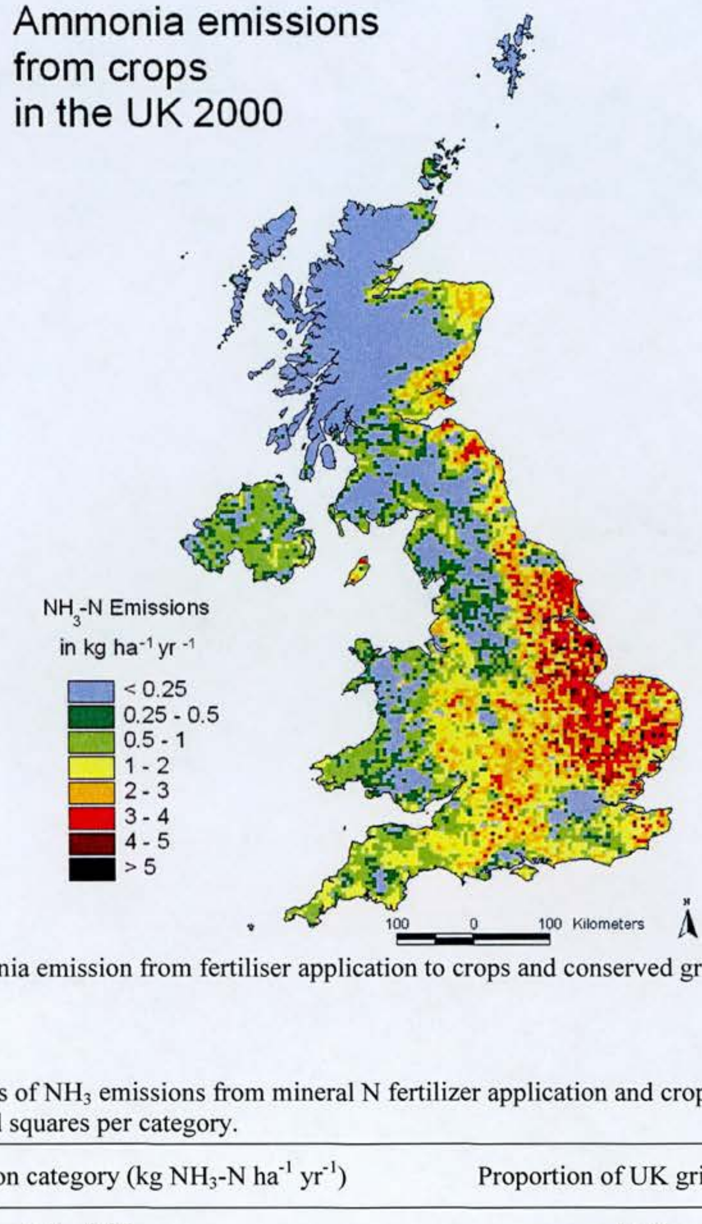

Figure 8.3. Ammonia emission from fertiliser application to crops and conserved grassland for UK in 2000.

Table 8.8. Analysis of NH_3 emissions from mineral N fertilizer application and crops for UK in 2000: % of 5 x 5 km grid squares per category.

Ammonia emission category (kg NH_3-N ha^{-1} yr^{-1})	Proportion of UK grid squares
> 0 – 0.25	37 %
> 0.25 – 0.5	11 %
> 0.5 – 1	18 %
> 1 – 2	16 %
> 2 – 3	11 %
> 3 – 4	6 %
> 4 – 5	1.3 %
> 5	0.2 %

8.3.3 Spatial distribution of ammonia emissions from non-agricultural sources

Non-agricultural sources have been estimated at 44.7 kt NH_3-N for 2002 (Dragosits *et al.*, 2004), and NAEI (www.naei.org) for some sources (e.g. industrial, combustion, transport etc.). This emission estimate is equivalent to about 16.8 % of the total ammonia emission in the UK. Non-agricultural emissions were not calculated independently in this study, and therefore the above emission estimate for 2002 was incorporated in the emission estimate for year 2000.

The spatial distribution of non-agricultural emissions 2002 is shown in Figure 8.4. Details on the distribution methodology for non-agricultural sources can be found in Section 3.7. The majority of the grid cells (26 %) were in the range of 0.5 – 1 kg NH_3-N ha^{-1} yr^{-1}), and only a small proportion (5.6 %) emit more than 5 kg NH_3-N ha^{-1} yr^{-1} (see Table 8.9).

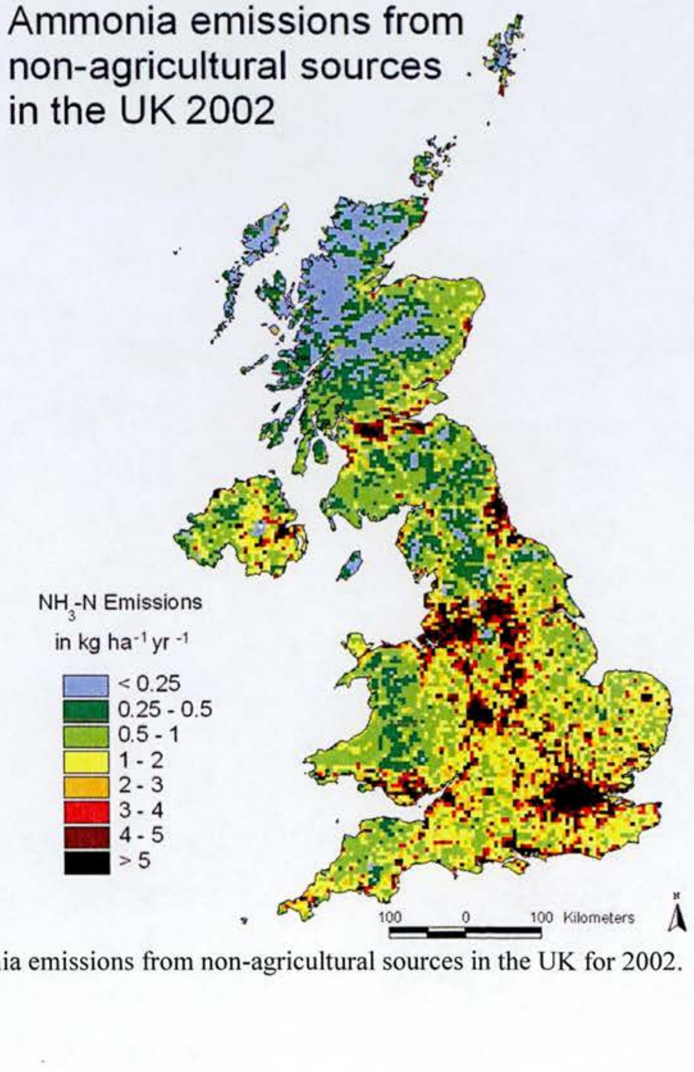

Figure 8.4. Ammonia emissions from non-agricultural sources in the UK for 2002.

Table 8.9. Analysis of NH_3 emissions from non-agricultural sources in UK for 2000: % of 5 x 5 km grid squares per category.

Ammonia emission category (kg NH_3-N ha^{-1} yr^{-1})	Proportion of UK grid squares
> 0 – 0.25	20 %
> 0.25 – 0.5	15 %
> 0.5 – 1	26 %
> 1 – 2	18 %
> 2 – 3	8 %
> 3 – 4	5 %
> 4 – 5	3 %
> 5	6 %

8.3.4 Spatial distribution of total ammonia emissions

The total ammonia emission for the UK in 2000, applying the AENEID model, was estimated at 265 kt NH_3-N. The contribution from each emission sectors is shown in Figure 8.5.

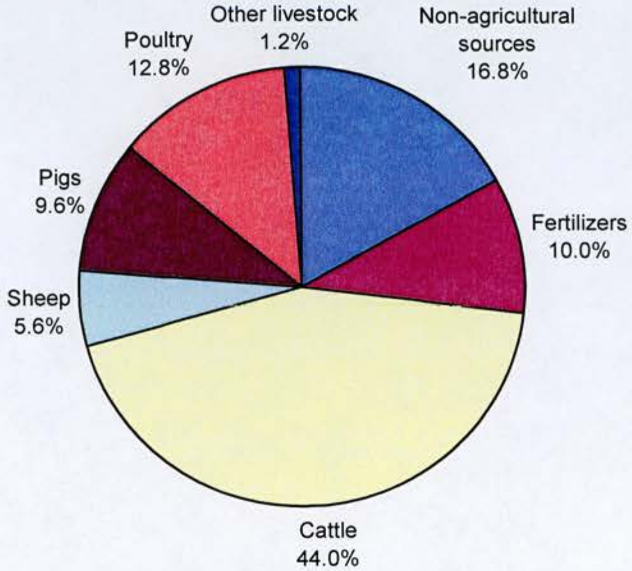

Figure 8.5 Emission contribution from the main emission sectors for 2000.

The emission estimates of the agricultural sources (Figure 8.2 and Figure 8.3) and the non-agricultural sources (Figure 8.4) were combined to derive the total ammonia emission map of the UK (Figure 8.6). The UK emission map for year 2000 (Figure 8.6) shows high emissions in Cheshire, Lancashire, Devon, Somerset, East Anglia, Yorkshire and Eden Valley and in some parts of Northern Ireland. These areas are

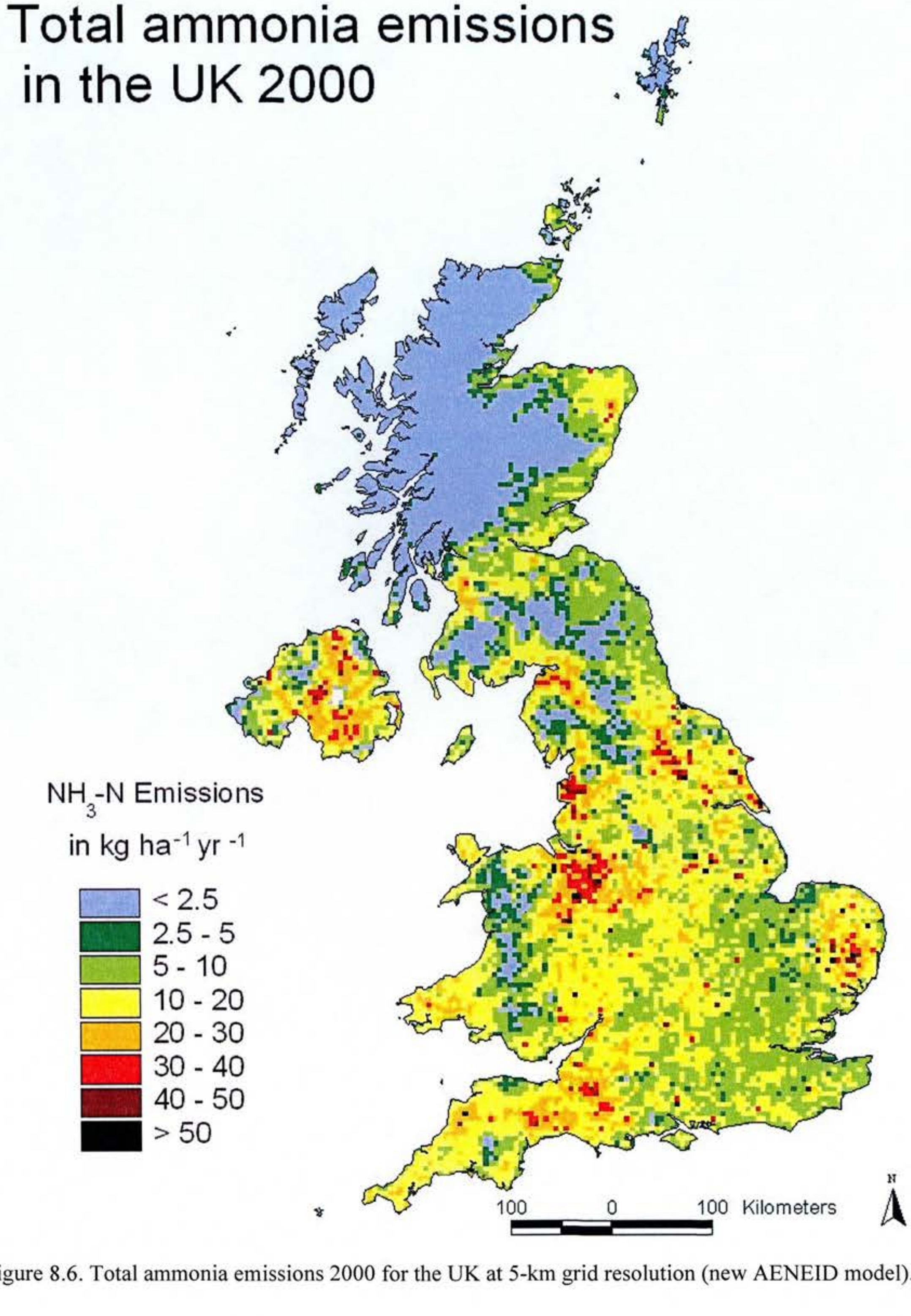

Figure 8.6. Total ammonia emissions 2000 for the UK at 5-km grid resolution (new AENEID model).

intensive agricultural areas dominated by livestock (dairy cattle, pigs and poultry) farming. The Scottish Highlands on the other hand, have relatively small emissions of ammonia. High emission values are common in areas with pig and/or poultry farming. Medium to high emission values are generally situated in dairy cattle dominated areas. Grid cells with low emission values are common in areas with sheep, beef or crop farming.

Figure 8.7 shows the contributions from agricultural and non-agricultural sources in each grid cell as a percentage of the total emission. The agricultural and non-agricultural maps show a complementary distribution pattern, as non-agricultural emissions mainly have been estimated to locations of low agricultural activity. A quantitative comparison of the two maps (Table 8.10), shows that agricultural sources are dominant (> 50 %) in 82 % of the grid cells, while non-agricultural sources are diminant in only 18 % of the grid cells.

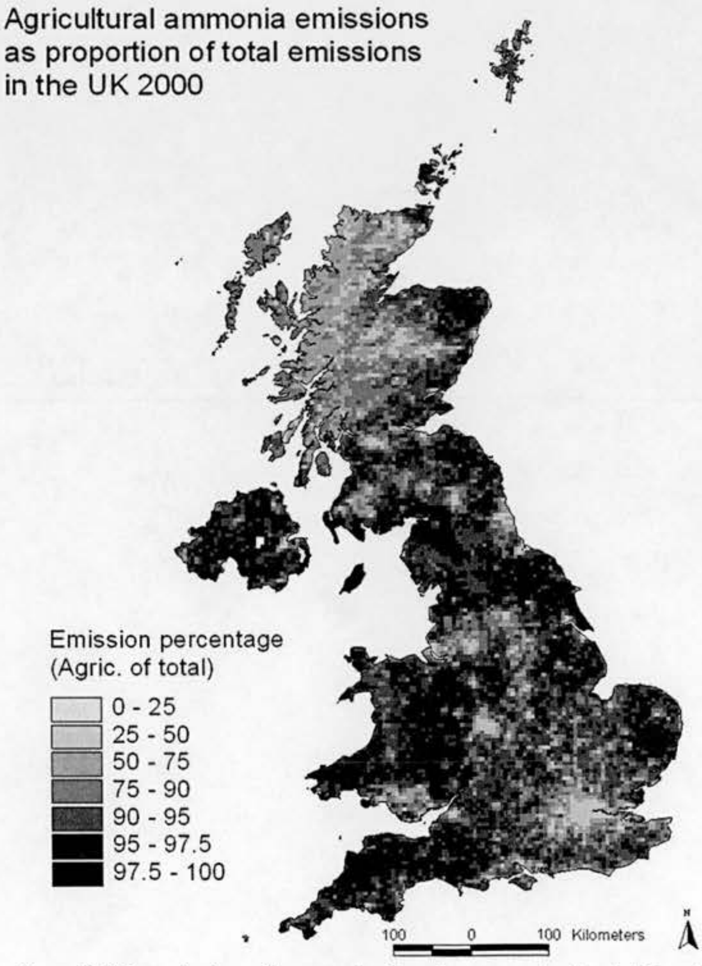

Figure 8.7. Contribution of NH_3 emissions from agricultural sources to the total emissions in the UK in 2000.

Table 8.10. Proportion of UK grid squares with % contribution of NH_3 emissions from agricultural and non-agricultural sources to the total emissions and of NH_3 emissions from livestock and fertilizer application to the total agricultural emissions in 2000.

Category (% contribution to total emissions)	**Non-agricultural sources** (proportion of UK squares)	**Fertilizer sources** (proportion of UK squares)	Category (% contribution to total emissions)	**Livestock sources** (proportion of UK squares)	**Agricultural sources** (proportion of UK squares)
0 – 2.5	1.3 %	24.9 %	0 – 25	9.2 %	6.2 %
2.5 – 5	11.2 %	25.4 %	25 – 50	17.8 %	11.9 %
5 – 10	21.9 %	18.8 %	50 – 75	30.6 %	19.4 %
10 – 25	28.1 %	20.3 %	75 – 90	31.8 %	28.1 %
25 – 50	19.4 %	8.5 %	90 – 95	10.2 %	21.9 %
50 – 75	11.9 %	1.9 %	95 – 97.5	0.3 %	11.2 %
75 – 100	6.2 %	0.2 %	97.7 – 100	0.0 %	1.3 %

8.3.5 Analysis of livestock sub-sources

Four sub-sources of livestock (cattle, sheep, pigs and poultry) for 2000 were analysed in detail. Figure 8.8 shows the contributions of different livestock types to the livestock ammonia emission.

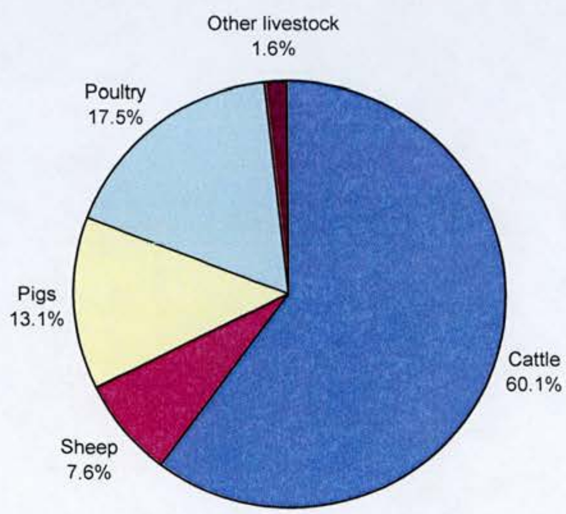

Figure 8.8 Contributions of different livestock types to the livestock ammonia emission from UK agriculture for 2000.

A map of dominant sources was calculated to analyse the spatial pattern of the main sub sources (Figure 8.9). A source was defined as being dominant if more than 45 % of the emission in that grid cell was represented by a single source. 45 % was considered suitable, as a cut-off value of 50 % would have resulted in many squares not being assigned a dominant source category. If no source represented more than 45 % of the emission, the grid cell was assigned to a "mixed" category. Grid cells with cattle as the dominant source were further sub-divided into dairy or beef, depending on the dominant cattle category in that grid cell. Grid cells with a total emission of less than 1 kg ha^{-1} were assigned to a "background" category.

Figure 8.9 shows that both cattle and sheep emissions tend to be more evenly distributed across the country than the more localised intensive pig and poultry sources. Emissions from poultry are generally situated in close proximity to population centres, whereas emissions from pigs are more regionally based. Sheep emissions are common in the upland and hill areas, whereas cattle emissions tend to be concentrated on lower-lying more fertile areas.

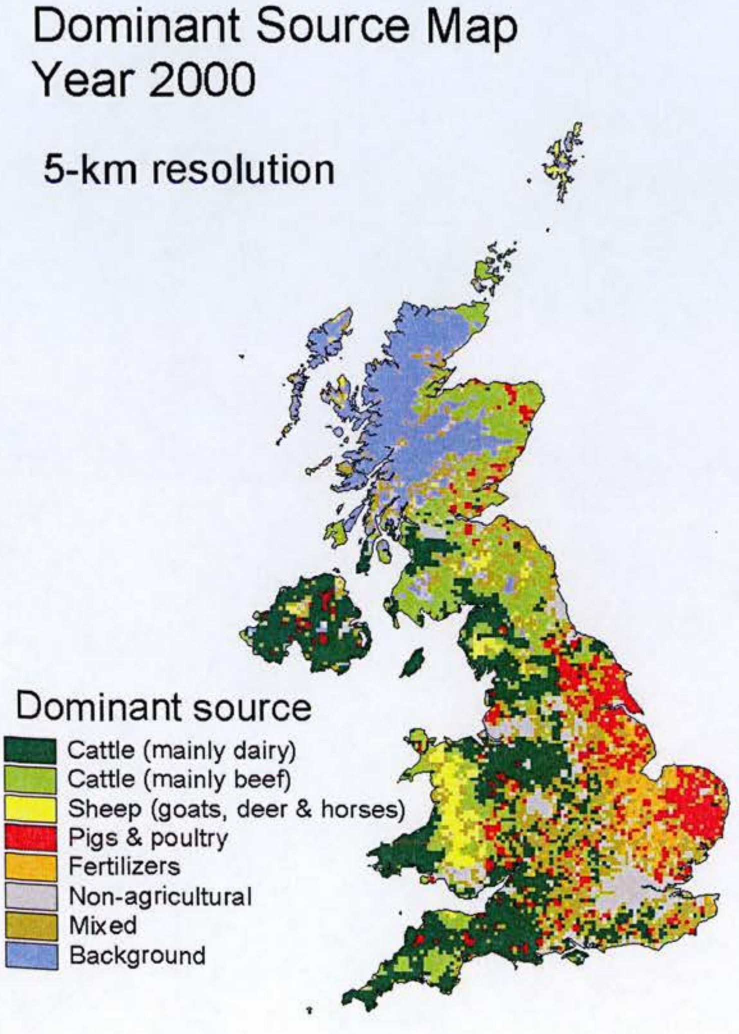

Figure 8.9 Spatial distribution of dominant sources of NH_3 emissions in the UK for 2000.

8.4 Comparison of the original and new AENEID model for livestock emissions in 1996

The ammonia emission map for 1996 based on the original AENEID approach by Dragosits (1999) was compared with the new emission map to assess differences between the two approaches, with main focus on livestock emissions. The main differences between the two approaches are in the emission potentials applied and in the modelling approach (with the implementation of the new sub-model to distribute spreading emissions from pig and poultry). Furthermore, the livestock categories were much less aggregated in the new approach (31 categories, Table 4.2) compared with the original approach (Table 8.11), where 11 livestock categories were spatially distributed, but only 5 emission potentials were applied.

Table 8.11. Categorisation of livestock categories in the original AENEID approach reagarding the spatial distribution and the emission potentials applied.

Spatial distribution categoriy	Emission potential
Dairy cows Other cattle	Cattle emission potential
Sheep & goats Lambs	Sheep emision potential
Sows Fattening Pigs	Pig emission potential
Laying hens Broilers Other poultry (ducks, geese, turkeys etc.)	Poultry emission potential
Horses	Horse emission potential

Emission potentials for the same livestock sectors (cattle, sheep, pigs and poultry) were derived from the new 1996 AENEID estimate by dividing the total livestock sector emission by number of livestock (Table 8.12). For instance, the total cattle emission for 1996 based on the new AENEID approach was estimated at 130.3 kt NH_3-N, and number of cattle was 11,894 thousand (Table 8.15). Hence, the cattle emission potential is 130.3/11.894 = 10.96 kg NH_3-N per cattle. The most significant difference in emission potentials in the old and the new AENEID approach occurs for poultry, where the emission potential is 35 % smaller in the new approach. The emission potential for cattle has also been reduced (by 11 %), while it has increased for sheep (3 %) and pigs (5 %). These changes are due to scientific advances and changes in management practice (e.g. combustion of poultry litter), improving the robustness of the emission potentials applied.

Table 8.12. Comparison of livestock numbers (GB) for 1996 and emission potentials applied in the original and new AENEID approach.

	Livestock numbers (1000's)			Emission potentials (kt NH_3-N animal^{-1} yr^{-1})		
	Original AENEID	New AENEID	Difference (%)	Original AENEID	New AENEID	Difference (%)
Cattle	10,169	10,228	+0.6 %	12.4	11.0	-11.3 %
Sheep	38,708	39,049	+0.9 %	0.34	0.35	+2.9 %
Pigs	6,908	6,924	+0.2 %	3.8	4.0	+5.3 %
Poultry	131,070	131,865	+0.6 %	0.34	0.22	-35.3 %

The original livestock AENEID emission map for GB 1996 was calculated by applying the emission potentials (Table 8.12) to the spatial distribution of the livestock categories (Table 8.11) based on the original AENEID approach (Section 3.5). The resulting map (Figure 8.10.a) could therefore be compared with the results of the new AENEID model for 1996 (Figure 8.10.c). An emission map based on the original AENEID distribution methodology, but applying the new updated emission potentials (Table 8.12) was also calculated (Figure 8.10.b) to assess the effect of the new distribution methodology.

The spatial distribution of ammonia emissions from livestock are similar in Figure 8.10.a and Figure 8.10.b, as they are both based on the same distribution methodology (the original AENEID approach). However, the overall emission in Figure 8.10.b is lower (184 kt NH_3-N) compared with Figure 8.10.a (211 kt NH_3-N) because the new updated emission potentials applied in (b) are generally lower than in the original estimate (Table 8.12). Comparing Figure 8.10.a and Figure 8.10.b therefore highlights differences due to the emission potentials applied. Figure 8.11.I and Figure 8.12.I show that livestock emissions are smaller applying the new emission potentials in most parts of the UK, with the exception of pig dominated areas due to the higher emission potential applied in the new approach.

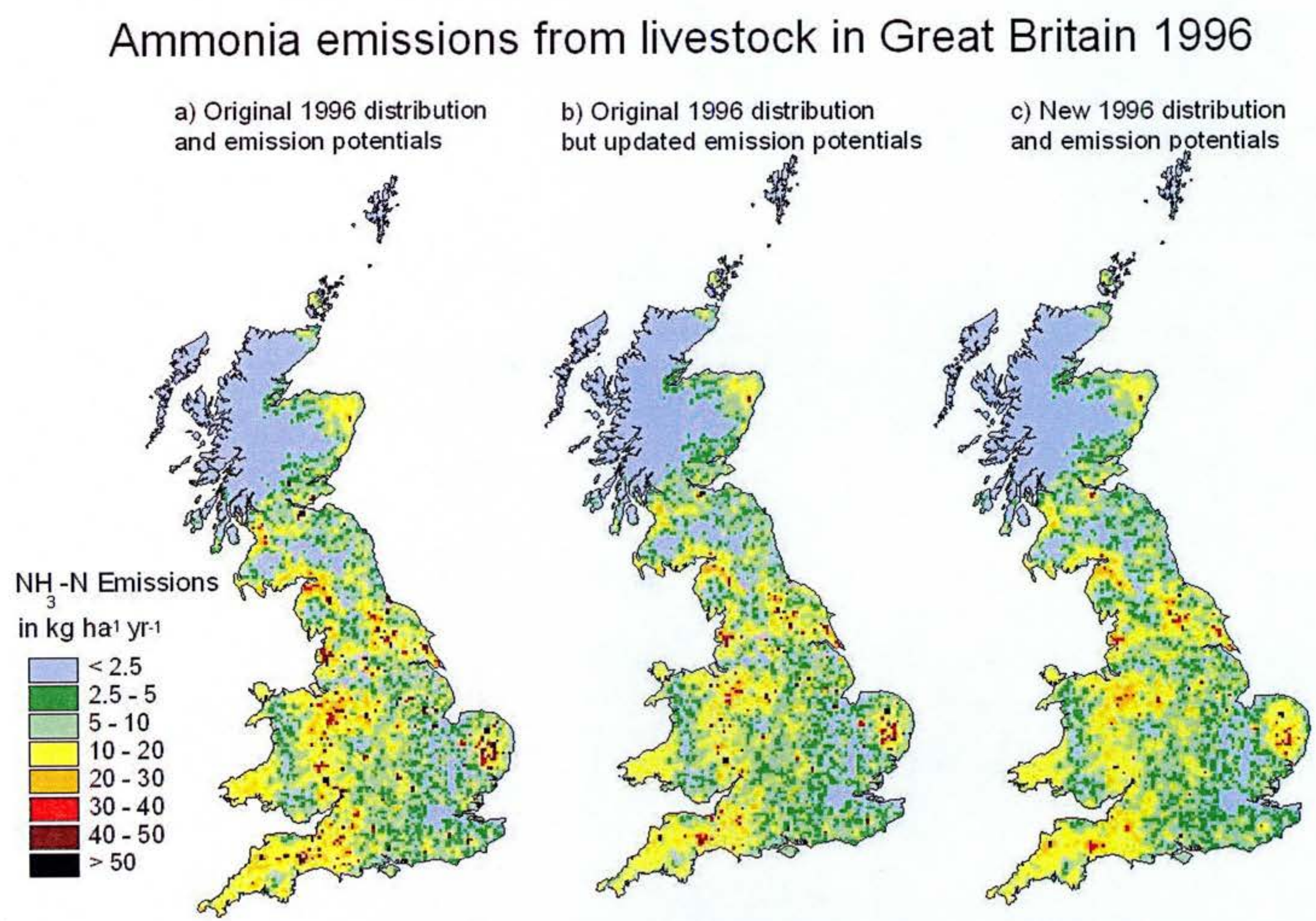

Figure 8.10. Ammonia emissions from livestock in Great Britain 1996 based on a) The original AENEID distribution methodology and original emission potentials, b) The original distribution methodology, but applying new updated emission potentials per livestock sector, c) The new AENEID distribution methodology and updated emission potentials per livestock category.

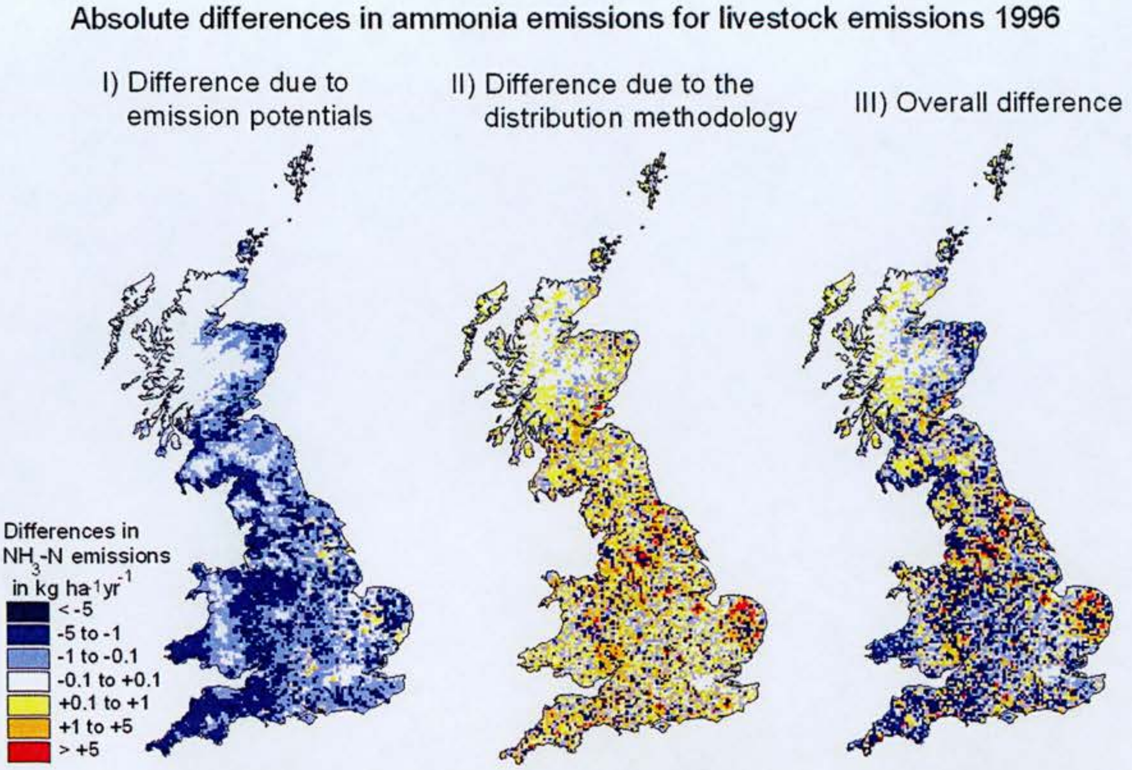

Figure 8.11. Absolute differences in ammonia emissions from livestock between the emission maps provided in Figure 8.10. I) shows the difference due to the emission potentials applied (Figure 8.10.b minus Figure 8.10.a); II) shows differences due to the distribution methodology and the agricultural census applied (Figure 8.10.c minus Figure 8.10.b); III) shows the overall difference (Figure 8.10.c minus Figure 8.10.a).

The total emission estimate in Figure 8.10.b (184 kt NH_3-N) is almost the same as in Figure 8.10.c (185 kt NH_3-N), as the same emission potentials have been applied. However, small differences occur due to the agricultural census data, where more animals are present in Figure 8.10.c (Table 8.12). These differences are fairly small (< 1 %), and hence, when comparing Figure 8.10.b and Figure 8.10.c, differences due to the spatial distribution methodology (original vs. new AENEID approach) are highlighted. From Figure 8.10 it can be seen that the new AENEID methodology provides less areas with high emission (Figure 8.10.b), due to the incorporated poultry model that distributes emissions from poultry manure over a greater area. For instance, in East Anglia, the number of very high emission areas (red areas) have been reduced, while medium to high emissions (yellow areas) have increased. This effect is also clearly shown in Figure 8.11.II and Figure 8.12.II.

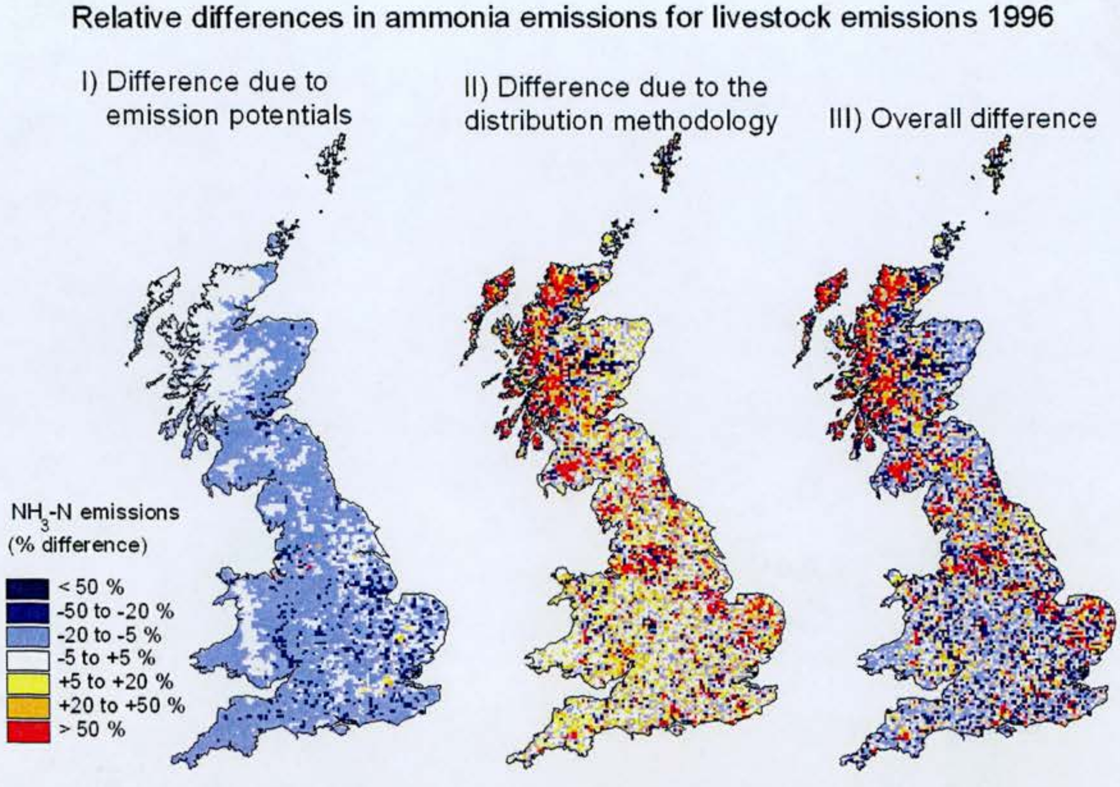

Figure 8.12. Relative differences in ammonia emissions from livestock between the emission maps provided in Figure 8.10. I) shows the difference due to the emission potentials applied (Figure 8.10.b minus Figure 8.10.a divided by Figure 8.10.a), II) shows differences due to the distribution methodology and the agricultural census applied (Figure 8.10.c minus Figure 8.10.b divided by Figure 8.10.b), III) shows the overall difference (Figure 8.10.c minus Figure 8.10.a divided by Figure 8.10.a).

The difference between Figure 8.10.a and Figure 8.10.c highlights the overall difference between the original and new AENEID livestock emission map for 1996. Figure 8.11.III and Figure 8.12.III shows that overall, the new AENEID emission estimate for livestock in 1996 is smaller in most parts of the UK (mainly due to decreased emission potentials applied). Higher emission areas occur mainly in pig and poultry dominated areas due to the re-location of pig and poultry emissions.

A quantitative comparison of emissions from 1996 based on the original and new AENEID approaches is provided in Table 8.13. Overall, the total agricultural NH_3 emission is 9 % smaller in the new emission inventory, compared with the original emission inventory for 1996. Livestock emissions have been estimated to be 12 % smaller, while fertilisers are estimated to be 28 % greater. Poultry emissions are significantly lower in the new approach (34 %) due to the smaller emission potential applied (related partly to increased litter combustion). Cattle emissions are also smaller (11 %) while both pig and sheep (including emissions from goats, horses and deer) emissions are higher (4 % and 13 % respectively). These changes are mainly due to changes in the emission potentials applied (Table 8.12).

Table 8.13. Comparison of total emission values in Great Britain for 1996 based on the original AENEID model and emission potentials derived from BBSRC (1997) and the new AENEID model and emission potentials derived from Misselbrook *et al.* (2004).

	1996 (original AENEID)		**1996 (current study)**		**Difference**
	Kt NH_3-N	% of total NH_3-N	Kt NH_3-N	% of total NH_3-N	(%)
Cattle	126	55.0%	112	53.8%	-11.1%
Sheep*	15	6.6%	17	8.2%	13.3%
Pigs	26	11.4%	27	13.0%	3.8%
Poultry	44	19.2%	29	13.9%	-34.1%
Total livestock	211	92.1%	185	88.9%	-12.3%
Total Fertilizer	18	7.9%	23	11.1%	27.8%
Total agriculture	229	100.0%	208	100.0%	-9.2%

* includes emissions from goats, horses & deer

The emission maps in Figure 8.10 were further assessed by scanning the emission values for each 5 x 5 km grid cell (Figure 8.13). These scanning series clearly show that high emission values have been reduced in the new AENEID methodology (c) compared with the original approach (a). Comparing scanning series (a) and (b) shows a similar spatial emission pattern, but generally lower emissions due to the lower emission potentials applied in (b). When comparing (b) and (c), it can be seen that the number and magnitude of emission peaks have been reduced in the new AENEID approach compared with the original methodology (due to the poultry sub-model).

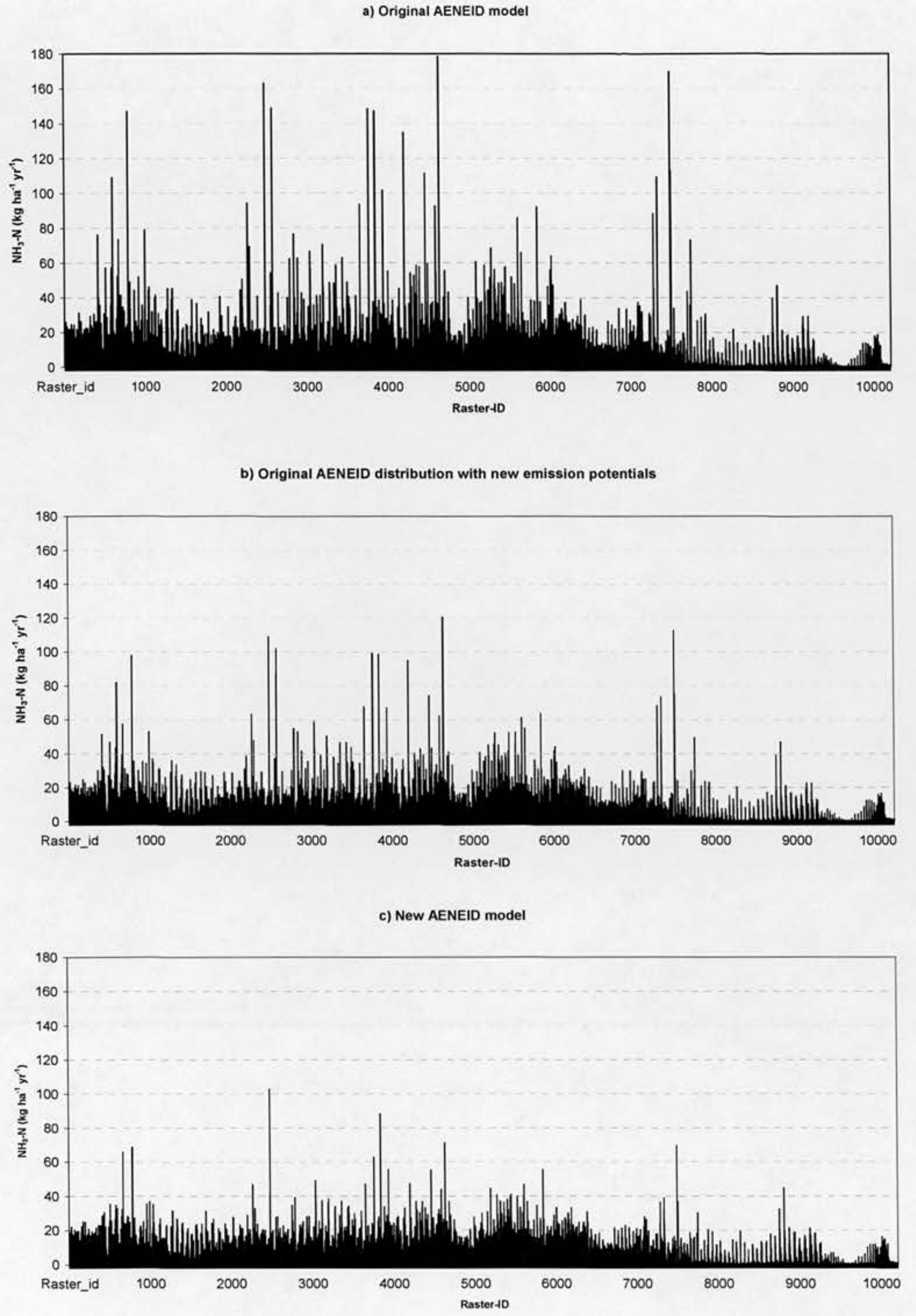

Figure 8.13. Scanning series of the emission values provided in Figure 8.10 where a) the original AENEID methodology and emission potentials were applied, b) the original AENEID methodology but new emission potentials, and c) the new AENEID distribution methodology and new emission potentials were applied.

A dominant livestock emission map was calculated for all three emission maps (Figure 8.14) to assess the importance of different livestock sources. These dominant source maps show grid cells where the main livestock source of NH_3 emission (> 45 %) derives from either cattle, sheep or pigs and poultry. In the new emission estimate, sheep dominated areas have increased due to the higher sheep emission potential applied. The number of grid cells dominated by pig and poultry are roughly the same in both approaches (15.4 % and 15.9 % respectively), however, the areas have shifted in some parts of the UK (for instance in East Anglia, where there is an increase in the area dominated by pig and poultry emissions), due to the new poultry sub model applied.

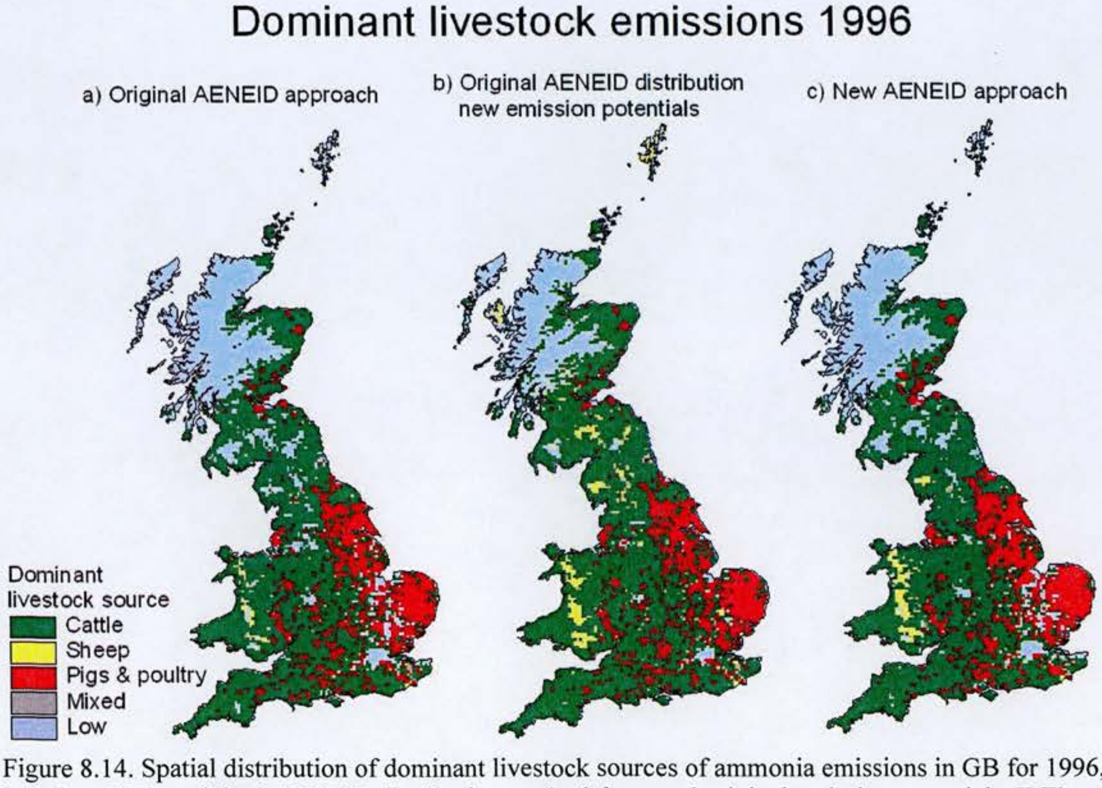

Figure 8.14. Spatial distribution of dominant livestock sources of ammonia emissions in GB for 1996, based on I) The original AENEID distribution methodology and original emission potentials, II The original distribution methodology, but applying new updated emission potentials per livestock sector (Table 8.12), c) The new AENEID distribution methodology and updated emission potentials per livestock category. (Low livestock emissions < 0.5 kg NH_3-N ha^{-1} yr^{-1}).

8.5 Verification of the AENEID model with the FRAME model and independent measurements

The atmospheric transport model FRAME (Fine Resolution AMmonia Exchange) incorporates the main atmospheric processes (emission, diffusion, chemistry and deposition) to calculate NH_3 concentration fields in the UK (Singles *et al.*, 1998; Fournier *et al.*, 2002; Fournier *et al.*, 2004; Vieno, 2006). The model incorporates horizontal and vertical gradients of NH_3 and calculates vertical concentration profiles with diffusion through 33 layers of varying depth.

A concentration field for ammonia, based on the new AENEID NH_3 emission inventory for 2000, was generated using the FRAME model (version 5.6). These modelled concentration fields (Figure 8.15) were compared with measured ammonia air concentrations in the National Ammonia Monitoring Network (NAMN) (Sutton *et al.*, 2001c; Tang and Sutton, 2004), see Section 1.5.4 to verify the results of the new AENEID model. Both the dispersion model outputs and the measurement data from the Network suggest that upland areas are characterised by small ammonia concentrations, while agricultural areas, particularly pig and poultry areas, are associated with large air concentrations of ammonia. The comparison suggested a fairly good fit of the magnitude and spatial variability of ammonia concentrations at a national scale, with an R^2 value of 0.6 (Figure 8.16). Table 8.14 suggests that emissions are overestimated by AENEID in cattle, pig and poultry dominated areas compared with the measurements in the monitoring network, while sheep and non-agricultural areas show a good correspondance. Comparing 5-km grid cell estimates from the FRAME model with point measurement data is likely to be associated with considerable scatter due to the smoothing effect of the 5-km estimate, as well as due to uncertainties and distance of the monitoring site in the network to local sources. This effect is particularly important in areas with high local variability in ammonia emissions, such as intensive agricultural areas.

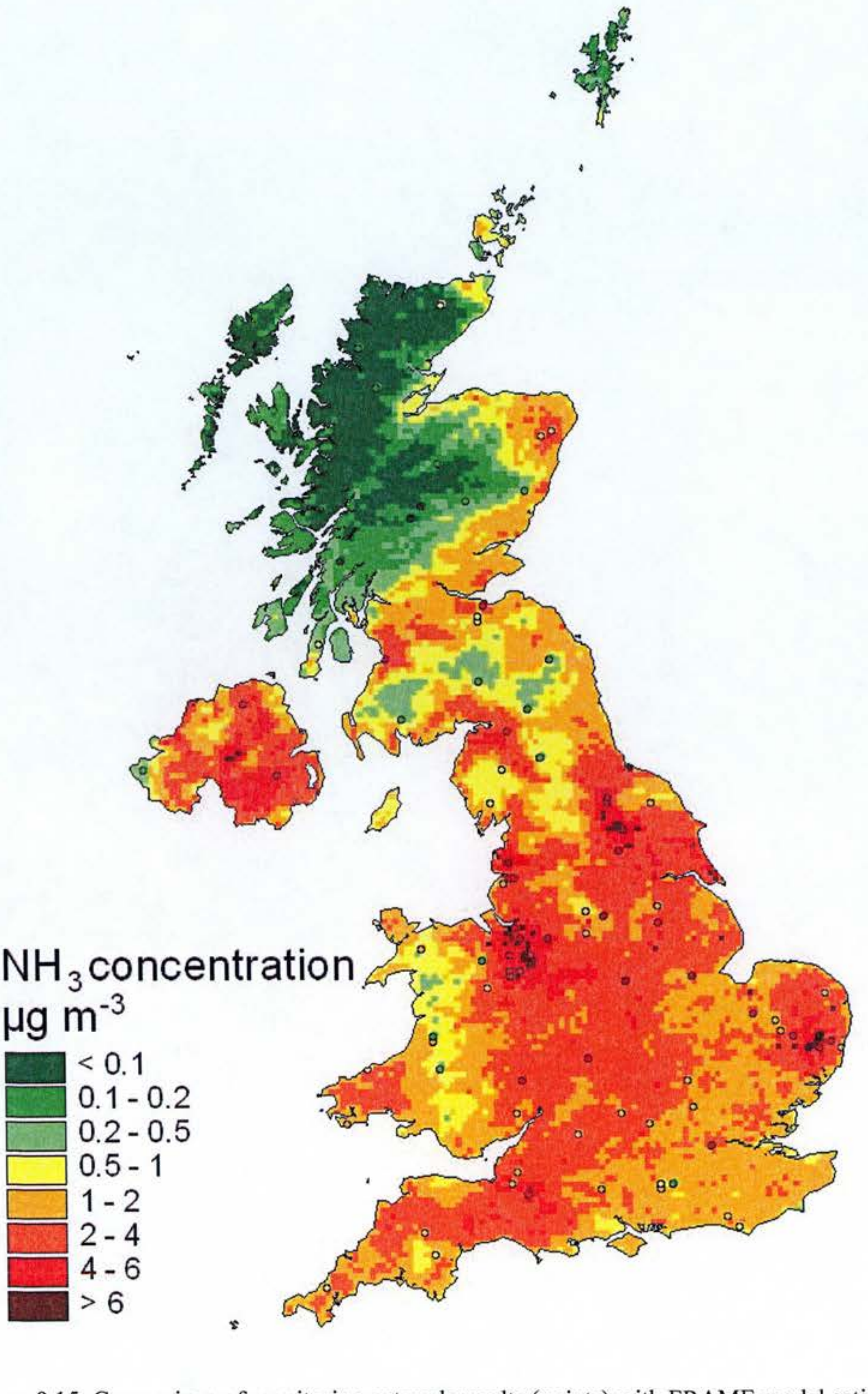

Figure 8.15. Comparison of monitoring network results (points) with FRAME model estimates using the 2000 ammonia emission inventory based on the new AENEID model.

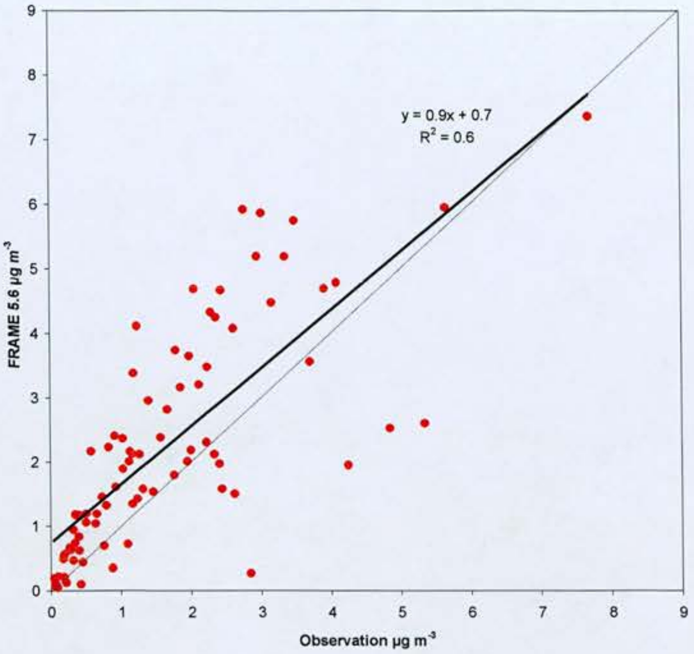

Figure 8.16. Comparison of concentration values of ammonia in the national ammonia monitoring network (NAMN) for year 2000 and corresponding 5 x 5 km FRAME model estimates.

Table 8.14. Paired comparison of average concentration values from the National Ammonia Monitoring Network (NAMN) for 2000 and the period with the results of the FRAME model for 5 km grid-squares surrounding the measurement sites using the new AENEID model estimates for 2000.

		NH_3 air concentration (µg m^{-3})		
Dominant source	No. of sites	AENEID-FRAME (2000)	NAMN (2000)	NAMN (1999–2003)
Cattle	34	2.64	1.66	1.87
Sheep	4	0.67	0.63	0.93
Pig & poultry	7	4.35	3.59	3.99
Non agricultural	10	1.42	1.52	1.51
Mixed	21	2.65	1.83	2.16
Background	11	0.28	0.19	0.30

8.6 Temporal changes in NH_3 emissions during 1990-2000

The ammonia emission trend during 1990 - 2003, as estimated by Misselbrook *et al.* (2004), is shown in Figure 1.15. These calculations show an overall gradual decrease in agricultural ammonia emissions from 318 kt NH_3 (262 NH_3-N) in 1990 to 248 kt NH_3 (204 NH_3-N) in 2003.

The total ammonia emissions for 1990, 1996 and 2000, as estimated using the AENEID model in this study, are summarised in Table 8.15. The agricultural ammonia emission for 1990 was estimated at 257 kt NH_3-N. The emissions are estimated to have declined to 234 kt NH_3-N for 1996, and decreased even further to 221 kt NH_3-N by 2000. Livestock emissions have declined mainly as a result of a declining number of animals (Figure 8.17), but also due to changes in management practice over the years (Section 8.2.1). Emissions from N fertilizer application for crop and grassland have also decreased from 1990, due to a general trend in less fertilizer N application and a smaller proportion of urea being used (Figure 8.18).

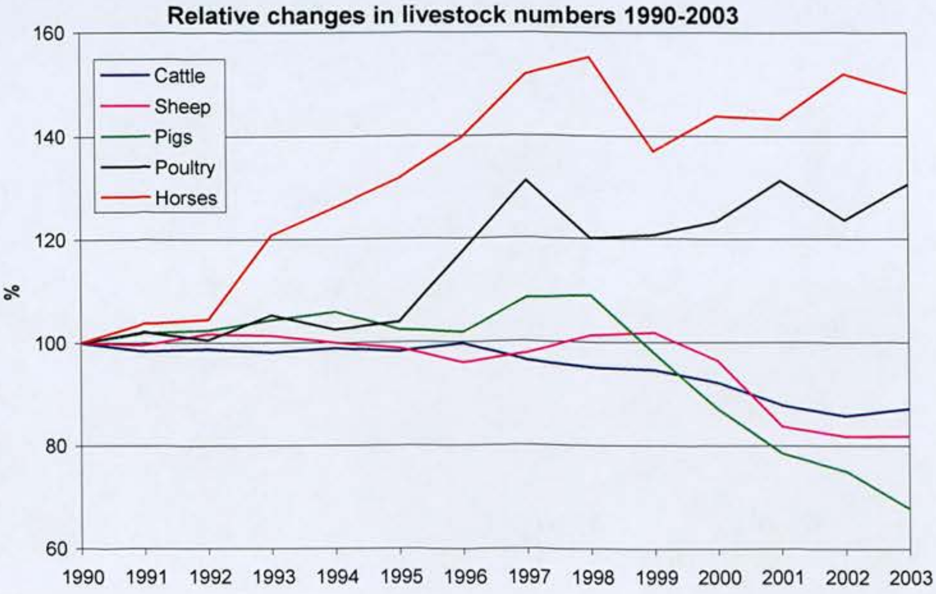

Figure 8.17 Trend in changes in livestock numbers since 1990. Source: Misselbrook *et al.* (2004).

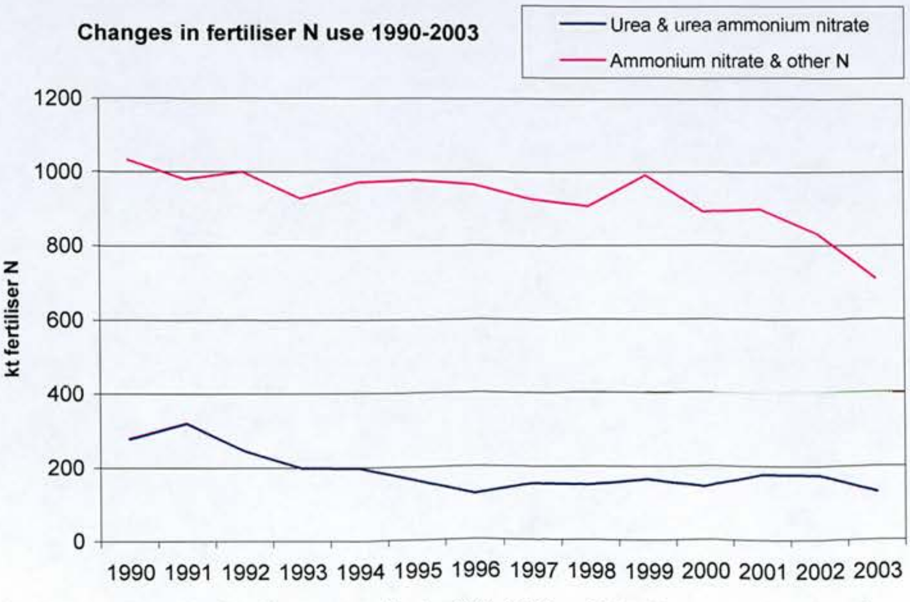

Figure 8.18 Trend in fertilizer N use in the UK 1990 – 2003. Source: Misselbrook *et al.* (2004).

Table 8.15. Estimated NH_3 emissions from agricultural sources in the UK 1990, 1996 and 2000 based on the new AENEID approach.

Category	Animals (1000's) UK 1990	Animals (1000's) UK 1996	Animals (1000's) UK 2000	Emissions (kt NH_3-N) UK 1990	Emissions (kt NH_3-N) UK 1996	Emissions (kt NH_3-N) UK 2000	Contribution % 1990	Contribution % 1996	Contribution % 2000
Cattle	11,925	11,894	10,903	135.2	130.3	116.7	51.2 %	55.7 %	52.9 %
Sheep	44,746	41,596	42,359	15.5	14.7	14.8	5.9 %	6.3 %	6.7 %
Pigs	8,172	7,490	6,477	32.9	29.6	25.5	12.5 %	12.6 %	11.6 %
Poultry	153,465	146,444	170,211	37.8	32.3	33.9	14.3 %	13.8 %	15.4 %
Other livestock				2.0	3.1	3.1	0.8 %	1.3 %	1.4 %
Total livestock				223.4	210.0	194.1	84.6 %	89.8 %	88.0 %
Fertilizer				40.6	23.8	26.5	15.4 %	10.2 %	12.0 %
Total agriculture				264.0	233.8	220.6	100 %	100 %	100 %

The spatial distribution of total ammonia emissions for 1990, 1996 and 2000 is shown in Figure 8.19.a-c. The emission total of these maps equals the emission total in Table 8.15. The spatial emission pattern in these maps is similar, as the same distribution approach (the new AENEID approach) has been applied. High emission levels tend to be representative in areas associated with intensive livestock farming, with the level of emission intensity changing slightly between the years. These differences are mainly due to changes in the agricultural census data, and to a smaller degree due to changes in the emission potentials applied

The same landcover data set was applied in all three distribution maps. The actual location of landcover types is not likely to change much during a 10 year period. It may be argued that LCMGB 1990 should be applied for the spatial distribution of ammonia sources for 1990. Fuller *et al.* (2003) however, do not recommend applying the two landcover datasets, LCM1990 and LCM2000, to estimate change over the 10 year period, because the two datasets are not directly comparable due to the different methods applied to generate them. The errors occurring due to the different data collection methods etc are likely to be larger that the actual landcover changes. Therefore, LCM2000 has been applied as input dataset for the calculation of ammonia emissions for all three years in this study (1990, 1996 and 2000) which provides a substantial improvement regarding accuracy, spatial extent and proportion of classified area, compared with LCMGB.

The same zonal aggregation data set were applied for all three distribution maps, with the exception of Northern Ireland, where the zonal system for 1990 was based on rural districts, compared with 5 x 5 km grid cells for 1996 and 2000. The spatial location of emissions is significantly influenced by the type of zone system applied for the aggregation of the agricultural census data as discussed in Chapter 9. When comparing the spatial distribution of emission in Northern Ireland for the three years, it can be seen that the emission pattern is smoother in 1990 compared with 1996 and 2000. This is to a large extent explained by the different zoning systems applied, rather than any relocation of the emission sources during the period.

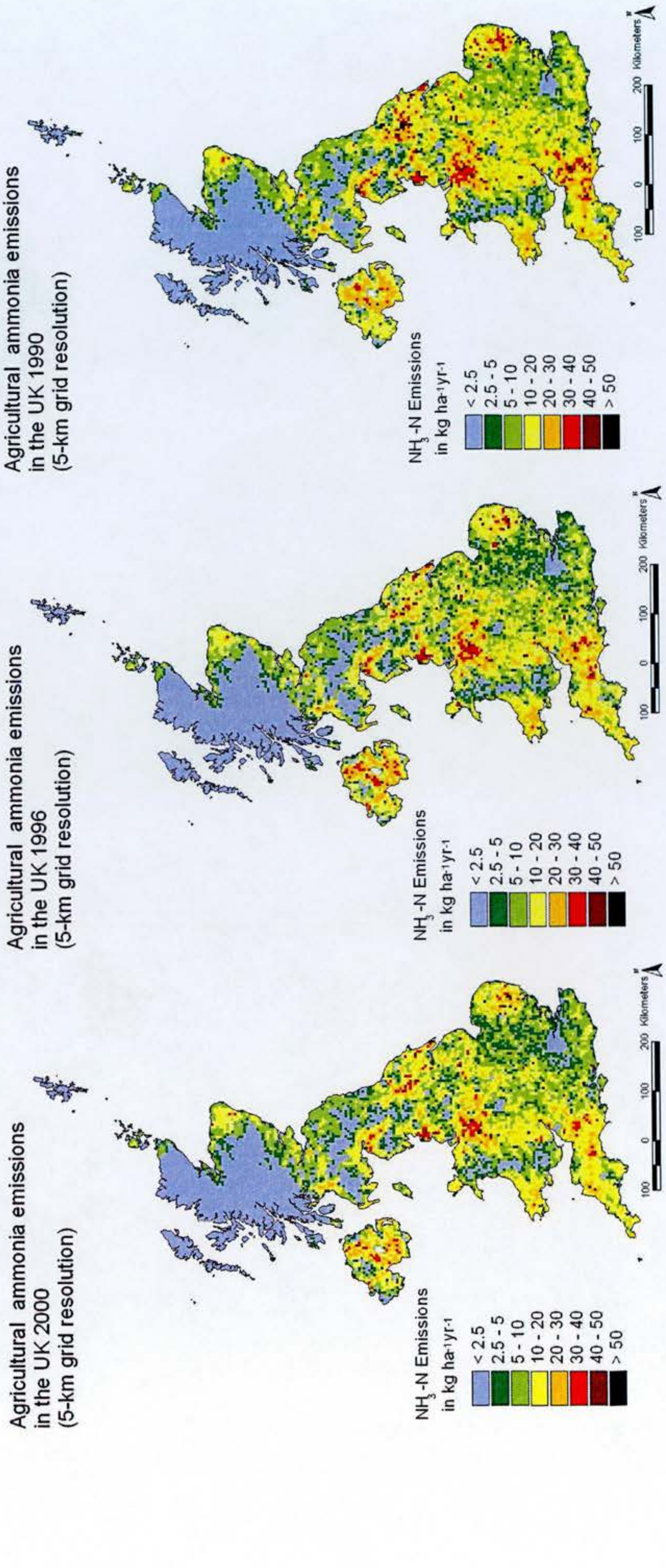

Figure 8.19.a-c. The spatial distribution of total agricultural NH_3 emissions in the UK a) 1990, b) 1996, and c) 2000, based on the new AENEID approach.

Figure 8.20 shows absolute and relative changes in emissions from agricultural sources in the UK during the study period (1990 to 2000). The emission has decreased in most parts of the UK. Areas that show an increase in emission (orange and red areas in Figure 8.20) were further analysed through comparison with the dominant source map (Figure 8.9). This comparison showed that the most significant emission increase has occurred in pig and poultry dominated areas, as a consequence of more pigs and poultry being reported in the agricultural census for these areas. The emission maps were also quantitatively analysed in detail (Table 8.16), to assess the extent of change that has occurred during the study period. About 80 % of the 5 x 5 km grid cells have remained at the same emission level or decreased in 2000 compared with 1990.

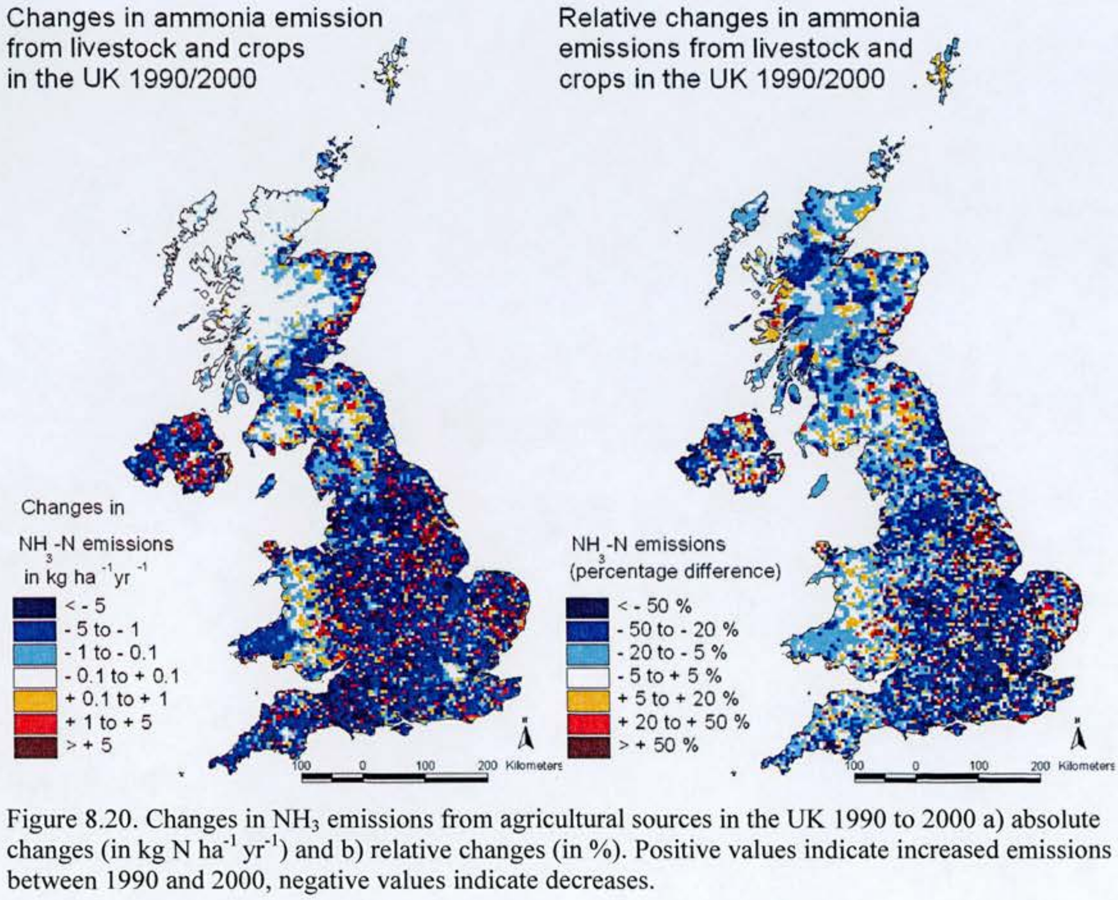

Figure 8.20. Changes in NH_3 emissions from agricultural sources in the UK 1990 to 2000 a) absolute changes (in kg N ha^{-1} yr^{-1}) and b) relative changes (in %). Positive values indicate increased emissions between 1990 and 2000, negative values indicate decreases.

Table 8.16. Difference in emissions patterns between 1990, 1996 and 2000 at 5-km resolution.

Difference	a) 1990 to 2000		b) 1990 to 1996		1996 to 2000	
(kg NH_3-N $ha^{-1}yr^{-1}$)	Area (km^2)	% of total area	Area (km^2)	% of total area	Area (km^2)	% of total area
50 to 300	25	0.01%	50	0.02%	0	0.00%
30 to 50	75	0.03%	0	0.00%	125	0.05%
20 to 30	150	0.06%	50	0.02%	175	0.06%
10 to 20	1,375	0.5%	475	0.2%	1,975	0.7%
5 to 10	4,000	1.5%	2,150	0.8%	5,275	1.9%
1 to 5	15,975	5.9%	10,450	3.8%	33,950	12.4%
0.1 to 1	18,000	6.6%	14,750	5.4%	40,425	14.8%
0.1 to – 0.1	61,400	22.5%	66,400	24.3%	80,000	29.3%
-0.1 to –1	54,250	19.9%	63,850	23.4%	53,000	19.4%
-1 to –5	87,025	31.9%	94,450	34.6%	50,300	18.4%
-5 to –10	23,525	8.6%	15,950	5.8%	6,075	2.2%
-10 to –20	5,500	2.0%	3,300	1.2%	1,125	0.4%
-20 to –30	850	0.31%	450	0.16%	125	0.05%
-30 to –50	350	0.13%	200	0.07%	100	0.04%
-50 to –300	225	0.08%	200	0.07%	100	0.04%

8.7 Summary and conclusions

The results of the new spatial AENEID NH_3 estimate of the UK show that high emissions are common in areas with intensive agricultural activity, such as intensive pig, poultry and dairy cattle areas. Livestock dominated areas are mainly situated in Northern Ireland, Wales, the western part of England and in south-eastern parts of Scotland, while fertilizer dominated areas are more common in eastern parts of England. Cattle are the dominant emission source in many lowland areas, especially in the western part of the UK. Emissions from sheep, on the other hand, dominate upland and hill areas. Emissions from pigs and poultry are more localised than cattle and sheep emissions. This is, however, less evident in the new AENEID approach, where these emissions are smoothed out over a larger area, compared with the original approach, due to the incorporation of a sub-model that distributes manure spreading emissions from poultry and pigs beyond the parish of origin (Section 7.4.1).

The spatial distribution of emissions for year 1996, applying the new AENEID approach, was compared with the results of Dragosits (1999), who applied the original AENEID approach based on BBSRC (1997). The main differences between the two approaches are the distribution methodology for pig and poultry sources, the landcover map applied to allocate NH_3 sources (LCM1990 compared with LCM2000) and the emission potentials applied (BBSRC (1997) compared with Misselbrook *et al.* (2004)). Furthermore, there are also differences in the agricultural statistics used, both regarding the census data and level of categorisation for livestock and crops (13 categories previously compared with 46 in the revised AENEID).

When comparing the agricultural emission results of the original AENEID model with the new approach, the new approach provides less extreme emission peaks, mainly as a consequence of the newly developed pig and poultry sub-model. Overall, the estimated agricultural emission for Great Britain 1996 was 9 % smaller applying the new AENEID approach based on the new updated emission potentials of Misselbrook *et al.* (2004). The new emission estimate is considered to be associated with less uncertainties, as new updated emission potentials have been applied.

When the results of the new AENEID model (year 2000) was evaluated with the FRAME model and independent measurements, the comparison suggested a fairly good fit of the magnitude and spatial variability of ammonia concentrations at a national scale (R^2 value of 0.6).

Ammonia emissions in the UK were also assessed regarding overall changes in emissions during the study period from 1990 to 2000. During this period, emissions decreased, mainly as a result of decreased numbers of animals, but also due to lower fertiliser application rates, and a smaller proportion of urea used.

9 Effects of the Modifiable Areal Unit Problem on spatial ammonia emission inventories

9.1 Introduction

It is commonly accepted that the main sources of uncertainty in spatial emission inventories are in the way models represent reality and the input data to such models. Sources of uncertainty in non-spatial emission inventories may be in the activity statistics (representing the polluting activity), the emission potentials (the emission estimate per unit of polluting activity) or in the modelling methodology. For many emission inventories, uncertainties in emission potentials and activity data have been estimated by identifying upper and lower limits of certainty (Sutton *et al.*, 1995; Kühlwein and Friedrich, 2000; Misselbrook *et al.*, 2000; Winiwarter and Rypdal, 2001). When emissions are spatially distributed, a further dimension of uncertainty is added (Lindley *et al.*, 2000). While uncertainties in the magnitude of emissions have generally been fairly well investigated, uncertainties due to spatial issues tend to have been overlooked in the past. Some recent studies have investigated spatial uncertainties in spatial emission inventories to some extent (Dragosits, 1999; Lindley *et al.*, 2000; Winiwarter *et al.*, 2003).

Ammonia emissions vary greatly at a local scale and effects (eutrophication, acidification) occur primarily close to sources. Thus it is important to minimize uncertainties in the spatial location of the estimated ammonia emissions, as such uncertainties will have an influence on the very localised deposition of NH_3 (Sutton *et al.*, 1998). Errors and uncertainties in an ammonia emission map will inevitably have implications on the result of models that use it as their main input data, e.g. atmospheric transport and deposition models and critical loads exceedance assessments.

The main source of ammonia emissions is agriculture, and agricultural census statistics are the most important input data to an ammonia emission inventory. In the UK, agricultural statistics are collected at farm level, but are aggregated to parish level or 5-km grid resolution for distribution to users. In this chapter, the Modifiable

Areal Unit Problem (MAUP), associated with such amalgamation, is investigated in the context of undertaking an ammonia emissions inventory.

9.1.1 MAUP and Agricultural Census Data

The Modifiable Areal Unit Problem (MAUP) arises when data collected on a micro scale are aggregated and treated as an individual unit (a zone) in an analysis (Openshaw, 1984). Most spatial emission inventories use zone-based data as input, and spatial NH_3 emission inventories are no exception. Agricultural census statistics in the UK are collected at farm/holding level, but aggregated to e.g. 5-km grid cells, civil parishes or parish groups to comply with confidentiality constraints, as well as ease of use of very large data sets. These aggregation zones have generally been accepted to provide a reasonable balance between spatial uncertainty and resolution in models (Asman *et al.*, 1998). There is, however, little knowledge of the actual effect of the resolution of the aggregation. When agricultural census data are aggregated from farm-level, the data are generalised and variability within each zone is lost. In addition, this loss of information is not necessarily consistent from one zone to the other (Openshaw and Rao, 1995). Aggregated data give different results depending on the scale, size, shape and location of the aggregation zones. This problem is referred to as the Modifiable Areal Unit Problem (MAUP). Although many studies on the effects of the MAUP with regard to survey research, epidemiology etc., can be found in literature (Curtis and MacPherson, 1996; Svancara *et al.*, 2002; Guo and Bhat, 2004), the present study appears to be the first research efforts demonstrating effects of the MAUP in the context of spatial emission inventories.

Aggregating the agricultural holding data into zones ensures that information about any one holding in the census results will not be identifiable in most cases. Additionally, a number of 'confidentiality mechanisms' (data modification), may be applied in order to ensure confidentiality (Rees and Martin, 2002), especially for parishes with very few farms or with rare livestock or crop types. Summarising the farm holding data for a parish may result in errors, because the boundaries of parishes and farms do not normally match and farm boundaries are not available at a national level. Geddes *et al.* (2003) suggest that geographical variation in the physical characteristics of the farms and the parishes is the most significant problem in spatial modelling of these types of data.

Point data (such as farm holdings) can be difficult to analyse, but when the data have been aggregated into zones, spatial analysis of the data becomes possible. Other advantages of aggregating the data are that geographical patterns are created, and the volume of the data is sometimes reduced (Openshaw and Alvanides, 1999). The main disadvantage is that information and spatial detail is lost in the aggregation process. In addition to this, uncertainty may be introduced to agricultural census data as a result of incorrect information given by the farmer, e.g. missing or imputed values. These errors are not studied here.

The term 'modifiable' refers to the 'number and geometric arrangement' of spatial units (the zones) can be changed, and a different distribution could be generated if a different zonation system was used. Altering the existing zonal boundaries, and/or changing the number and hence the size of the zones affect the result of studies where zonal aggregated data are used. Aggregation of the data can be achieved in many different ways, both scale wise and zone wise (Openshaw, 1977). MAUP arises due to two effects (Openshaw and Taylor, 1979; Openshaw, 1984):

- *The scale effect* – the same data may give different results depending on the number of zones used.
- *The zonation effect* - results may vary even when the same number of zones are used, depending on how the boundaries of the zones are drawn.

Census geography refers to the zones (aggregate areas) for which the agricultural census data are published (Mackaness and Towers, 2002). There are in fact four different census geographies for the United Kingdom, as Scotland, Wales, England and Northern Ireland all use different systems for aggregating and distributing agricultural census data.

9.1.2 Modelling ammonia emissions

As discussed earlier, the general methodology to model ammonia emissions is to multiply an emission potential with spatially distributed activity data such as the agricultural census statistics. Ammonia emissions are modelled at a 1-km grid resolution in this study. The agricultural census data for England are normally aggregated and made available at parish level or 5 x 5 km grid level, and therefore have to be downscaled to the 1-km grid modelling resolution. This downscaling makes it difficult to estimate the "true location" of the census item within the aggregation zone. While it is technically possible to aggregate small units into larger units (up-scaling), down-scaling is not possible without additional information (Montello, 2001). When the agricultural census data for each zone (parish or 5-km grid cell) are re-distributed at the 1-km grid, a spatial representation error is introduced. The magnitude of the error depends on the zone size, as well as the location of zonal boundaries (Longley and Batty, 1996). Data can be re-distributed within the zone using supplementary data, i.e. by 'intelligent area weighted interpolation' (Sadahiro, 2000). This approach has been used in the past to reallocate the census items within each parish at a 1-km grid using landcover data in the AENEID approach (Dragosits *et al.*, 1998). Introducing a geographical property such as landcover data within the parishes is a means to reduce the spatial representation error within each zone because ammonia emissions from different agricultural sources tend to occur on specific landcover types. Landcover is a geographical property that correlates well with most agricultural data (except non-landbased enterprises such as large intensive pig and poultry farms).

9.2 Methodology

In this study, confidential farm holding data for England have been obtained and analysed. Due to the nature of the data, confidentiality issues somewhat limit the extent to which they can be visualised. All figures showing emission calculations are based on true data, and not considered to violate the confidentiality of the farmers. All figures representing holding data on the other hand, contain additional random data points, thereby ensuring the confidentiality of the data.

The holding data are well suited to analyse the MAUP, because they provide the opportunity to investigate the scale and zoning effects associated with different aggregations of census data. The MAUP and its effects are thus investigated by aggregation of (point) holding data for England using different zoning systems (see Figure 9.1). Four different zoning systems are tested here. Three gridded zoning

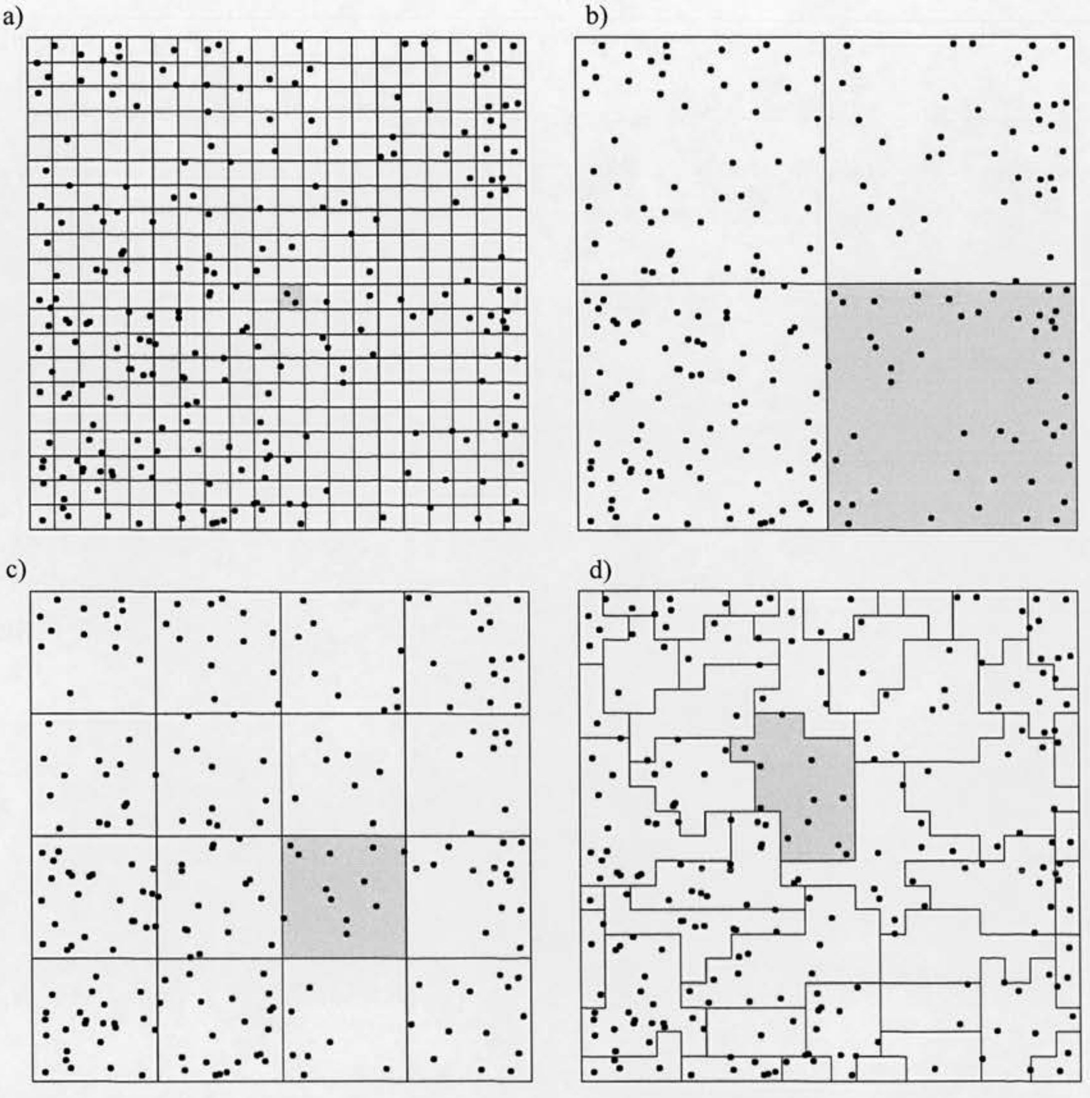

Figure 9.1. Holding data were aggregated using four different zonal systems: a) 1-km grid, b) 10-km grid, c) 5-km grid, and d) parish zones. The highlighted zones in each figure show the potential emission area at each level for a farm located in the 1-km grid cell of Figure a (also see Figure 9.2).

systems (1-km, 5-km and 10-km level) were chosen because a regular square pattern facilitates further analysis of the data and many data sets are made available aggregated to regular grid data. The fourth zoning system uses civil parishes, because this is the aggregation format usually available to users. Initially, all different census items (approximately 100) in the agricultural census data were aggregated into 46 source categories (Table 4.2). This aggregation is based on a set of livestock and crop categories for which separate NH_3 emission potentials are available, and is suited to link together the diverse datasets of the different devolved regions of the UK for a national NH_3 emission inventory. Each point (holding) was assigned to a zone in the four zoning systems. Agricultural census data for all holdings in each zone were then aggregated and then each NH_3 source category was re-distributed within each zone at a 1 x 1 km grid according to the AENEID methodology. Ammonia emission maps for each zonal system were calculated at a 1 x 1 km grid by applying emission potentials to the distributed source activity maps. The same original activity data and emission potentials were applied to all four scenarios, the only difference being the type of zoning system applied. It follows that, since the four scenarios are based on different zonal systems, for the source area extents, any spatial differences in the location of the emissions will be indicative of the effect of the MAUP.

An emission map based on pre-treated non-disclosive agricultural census data was also calculated and compared with the other emission maps. These data were obtained from the Edinburgh University Data Library (EUDL) and were already spatially distributed at 2-km grid resolution Similar to the AENEID approach, the parish data had been redistributed on a 1-km grid using land cover data, but the allocation rules applied (EUDL, 2004) were different from those used in the AENEID approach. The EUDL approach re-distributed the agricultural census items as such, while the AENEID approach re-distributes the census items as ammonia sources. Another difference between the two methods is that the EUDL approach re-distributes the census items evenly onto all land that is not excluded from agricultural use (e.g. freshwater and urban), rather than distinguishing between different levels of intensity of agricultural land as in the AENEID approach. This is due to the EUDL landcover data containing only one of seven landcover types per grid cell, whereas the landcover data used in this study (LCM2000, Fuller *et al.* (2002)) contain percentages of 26 different categories. This results in each 1-km grid cell in the EUDL-approach being 100 % suitable or unsuitable, while the grid cells in the AENEID approach can be anything from zero to 100 % suitable. The EUDL approach also applies other additional processing in order to ensure confidentiality in the non-disclosive data set and was finally aggregated to 2-km grid resolution.

The potential areal extent of ammonia emissions from a single holding is demonstrated in Figure 9.2. The areal extent and the spatial location of the emission depend on the size of the zones (the scale effect), as well as the location of the zones (the zonal effect). Large zones tend to dilute the emission over a larger area, whereas small zones may concentrate the emissions. The scale effect can be seen in Figure 9.2. when comparing the areal extent of the three grids (1-km, 5-km & 10-km). Although the areal extent of the sample parish and the 5-km grid in Figure 9.2 are almost of the same size, the location attributed to the emission source is very different because of the location of the zones, demonstrating the zonation effect.

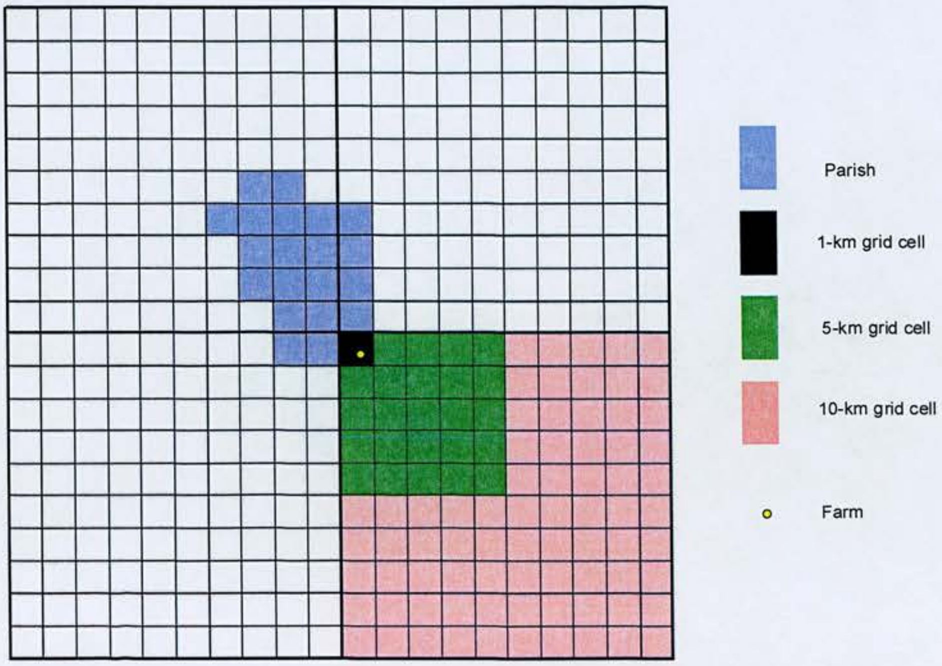

Figure 9.2. Area for re-distribution from a single farm to different aggregation zones (10-km grid, 5-km grid, 1-km grid and parish zones). Both the scale and the zonation effect are demonstrated.

When emission maps for all four aggregation levels, including the non-disclosive EUDL 2 x 2 km emission map, had been calculated for England, the gridded emission maps were analysed and compared. Winiwarter *et al.* (2003) suggest several different methods to compare gridded emission maps:

- *Visual interpretation* - Emission maps can be compared visually to identify differences in the spatial pattern. A disadvantage is that the number of values that can visually be distinguished is small since the eyes' interpretation of colour is limited.
- *Extreme emission values* – Extreme emission values for all emission maps can be identified and analysed.

- *Scanning data series* – The emission values for each 1-km grid cell can be scanned from left to right, line by line from the bottom to the top of a gridded dataset, to compare absolute numbers rather than colour ranges, as is the case of emission maps (Winiwarter *et al.*, 2003). This makes it easy to detect extreme emission values, and their distribution within the grid.

The distribution of agricultural land (arable land and grassland) in the census data was also compared with LCM2000 to further evaluate the effect of the MAUP. The following methodologies were applied to compare the two data sets:

- *Scattergrams* – Two datasets can be compared by plotting the comparison parameter for corresponding grid cells as an x-y plot (Winiwarter *et al.*, 2003). Arable land and grassland in the land cover map (LCM2000) were plotted on the x-axis, and compared with the estimates of agricultural land in the census data at the different aggregation levels (1-km, 5-km, 10-km zones and parishes), plotted on the y-axis.
- *The coefficient of correlation (R^2)* – R^2 provides information on how well two datasets fit. R^2 may vary from zero to one, and the higher the value, the better is the correlation. The coefficient of correlation has the advantage of reducing the comparison to a single parameter (Winiwarter *et al.*, 2003).
- *The acceptability criterion* - The percentage of grid cells that fall within acceptable absolute thresholds defined by the user is calculated and the overall 'acceptability' of the comparison may be expressed as the percentage of data pairs that are acceptable as a proportion of all data pairs (Winiwarter *et al.*, 2003). In this study, the agricultural land recorded in the census was spatially distributed using AENEID and subsequently compared with LCM2000. A difference between the two datasets of 25 ha per 1 x 1 km grid (100 ha) was deemed as "acceptable", because 25 % over- or under- estimation was regarded as a suitable range to eliminate major errors.

9.3 Results and discussion

In this study, the availability of disclosive agricultural census data for individual farm holdings in England allowed the investigation of the MAUP effect on spatially distributed ammonia emissions. Normally agricultural statistics are not available at this level of detail, with more common census geographies being parishes (Scotland), grouped parishes (Wales) or 5 x 5 km grids (Northern Ireland and England). These zoning systems do not provide any options of scales and zoning method, and users are limited to these fixed census geographies. Studying the original holding data however, makes it possible to investigate the uncertainties that arise in the aggregated data sets due to the MAUP. Although the MAUP cannot be eliminated in spatial emission inventories, at least this study can provide more understanding of the uncertainties and suggestions on how to reduce some of them.

9.3.1 Visual interpretation

Emission maps at a 1-km grid resolution derived from different aggregation zones for an area of 20 km x 20 km are shown in Figure 9.3.a and emission maps for the whole of England are shown in Figure 9.4.a. At the 1-km resolution, the emissions are very scattered, and the emission map is characterised by high emission peaks, as well as many grid squares of zero emission. The 1-km emission map gives the impression of lower total ammonia emissions than the other maps, because of the amount of grid cells with zero emissions, even though the overall total emission is the same for all emission maps. The difference is that the emissions are concentrated into a smaller number of grid cells, because ammonia emissions are only calculated for grid squares containing a holding. Intuitively it can be concluded that this is not a very realistic representation of ammonia emissions for an agricultural landscape. The problem with agricultural census data is that, although it is collected at holding level (point source), the actual livestock and the agricultural land in reality represent an area source. It is not very likely that all livestock, pastures and cropland reported for a farm are located in the 1 x 1 km grid cell where the farm is registered. However, a possible solution could be to apply an approach that is similar to the poultry model developed in this thesis (see Section 7.4.1), where the agricultural statistics are distributed in a zone around the farm holding, rather than in the aggregation zone of origin.

Both the 5-km level and the 10-km level emission maps show a smoother emission pattern than the 1-km level map. Many studies of the MAUP have shown that larger aggregation zones tend to have a "smoothing effect" on the result, while smaller zones exaggerate local differences. Jelinski and Wu (1996) showed that information on spatial heterogeneity is lost or distorted in the aggregation process and that the loss of detail increases as the zone size increases. Larger aggregation zones, however, have the advantage of giving more statistically stable values. It is not only the size of the aggregation zones that affects statistical parameters: Jelinski and Wu (1996) also showed that, although the size of the zones remain the same, the variance changes even when the orientation of the zones is changed (the zoning problem). Even at the national scale (Figure 9.4), clear differences between the maps can be detected, although the general spatial pattern in the maps is the same. Again, it is the 1-km level emission map that stands out compared with the other results.

In both Figure 9.3 and Figure 9.4, the underlying aggregation zones were visible in the grid-based zoning systems even at the 10-km level. The emission map based on the parish distribution did not give rise to any visible artificial boundaries. This may give the impression that the underlying distribution zones are undetectable. Although the parish boundaries are as artificial as squares, they are by no means as easily detectable as squares. Parishes with varying sizes and shapes are more difficult to detect compared with a regularly gridded pattern. When superimposing the parish borders onto the parish distributed map, some of the parishes could actually be identified in the underlying emission map. This effect varies with location and size of the aggregation zones. For instance, pig and poultry emissions are often located in small parishes in lowland areas, while cattle and sheep emissions tend to be located in larger parishes in upland areas, and hence the underlying aggregation zones are more difficult to detect, as shown in Figure 3.4.

The emission map based on non-disclosive 2-km grid data was also compared with the emission results produced from the confidential data at different aggregation levels. Figure 9.3.e shows that the emission map based on the non-disclosive data significantly differs from the emission maps calculated from the confidential data, showing an even smoother emission pattern.

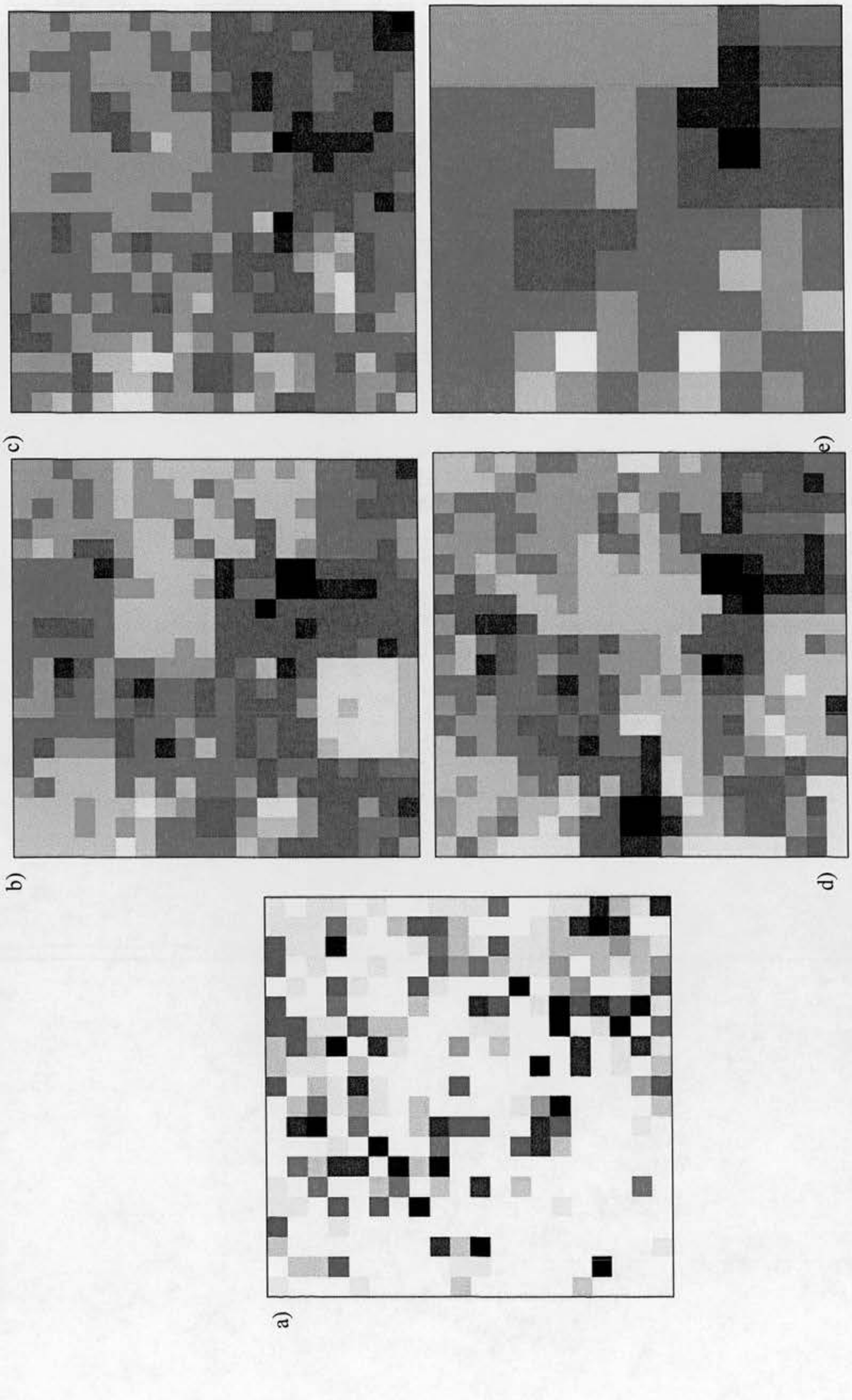

Figure 9.3. Examples of emission results (livestock emissions) at a local scale (400 km^2) at 1 km grid resolution using different aggregation zones. a) 1-km zone, b) 5-km zone, c) 10-km zone, d) parish zone, e) emission results from non-disclosive 2 x 2 km data. Note: These emissions are not based on the holding data in Figure 9.1.

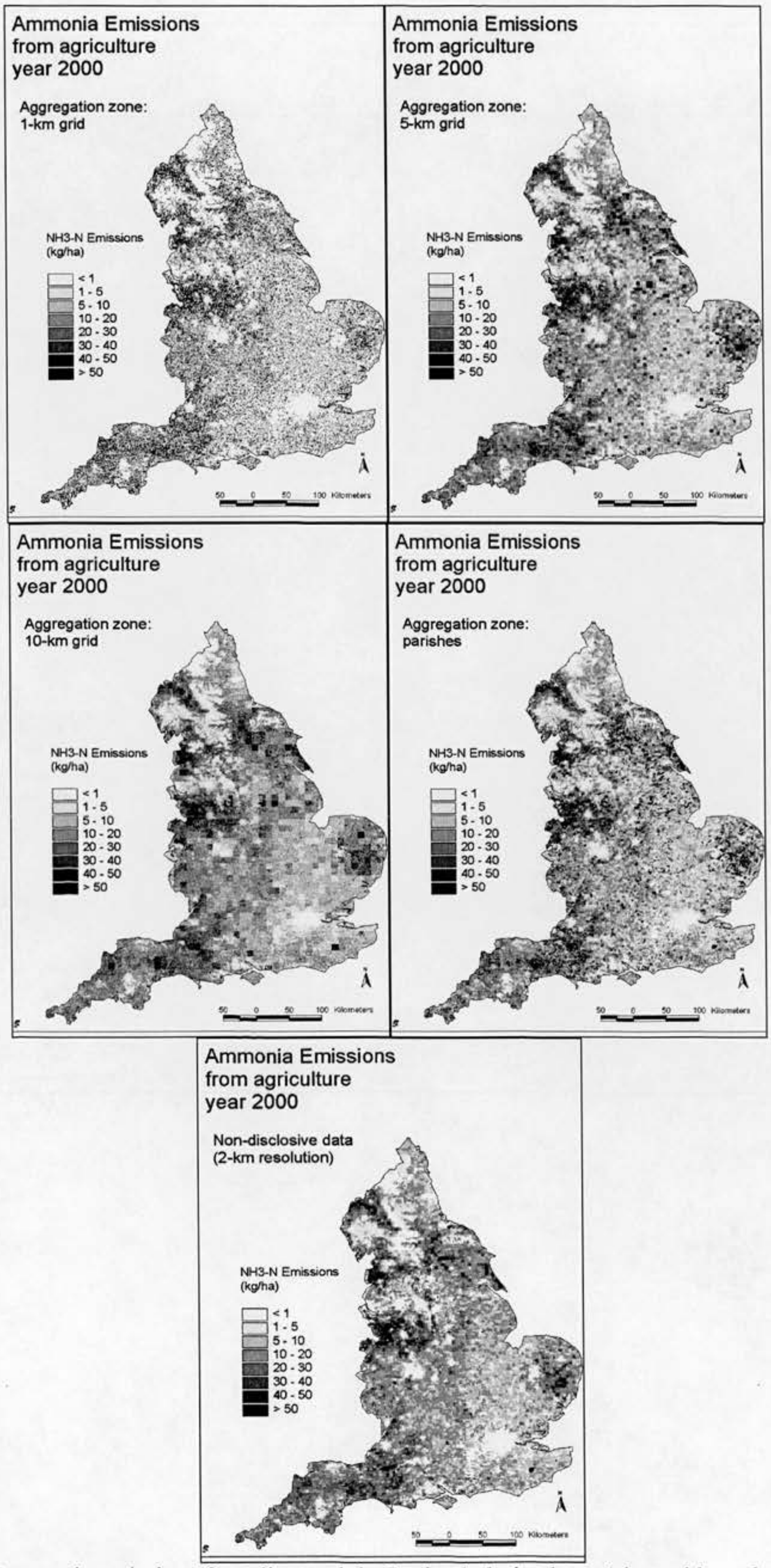

Figure 9.4. Ammonia emissions from livestock in England, derived at a 1 km grid resolution from agricultural census data aggregated at different zonal systems: a) 1-km zone, b) 5-km zone, c) 10-km zone, d) parish zone, e) emission results from non-disclosive 2 x 2 km data.

9.3.2 Extreme emission values

In Table 9.1, "extreme emission values" for each of the four aggregation levels were identified. As expected, the 1-km level has the most extreme values. It is followed by the parish distribution, the 5-km distribution, the non-disclosive map and finally the 10-km distribution. Table 9.1 clearly shows that emission values are less extreme the larger the aggregation zones (due to the smoothing effect).

Table 9.1. Maximum ammonia emission (tonnes NH_3–N $year^{-1}$) per 1-km grid square in the different emission maps.

Emission (t NH_3–N $year^{-1}$)	Aggr.zone: 1-km grid	Aggr.zone: 5-km grid	Aggr.zone: 10-km grid	Aggr.zone: Parish	Non-disclosive
Livestock	791.7	59.9	30.2	476.8	34.3
Fertilizers	21.2	2.1	1.2	5.9	2.0

It could be argued that there is nothing wrong with using the 1-km data approach to source area localisation, however, it is not suited to the AENEID approach, which is based on distributing emissions within an area rather than outside of it. Therefore, for 1-km zones, a new model more suitable for the data would need to be developed, focusing on distributing the agricultural statistics outside the aggregation zones, similar to the approach of the poultry model developed in Chapter 7.

Extreme values were common in the ammonia emission map based on the parish distribution. They tend to occur in small parishes, but high emission values may also be present in large parishes if the land cover type suitable for distributing agricultural emissions in that parish is limited to very few grid cells. If suitable land cover is not present in a parish, the AENEID model reassigns the census data to the next most likely land cover type. This is to ensure that no emissions are lost in the modelling process due to discrepancies between the census data and the landcover map. Despite this precaution, it was not possible to allocate a small proportion of the data to other appropriate zones using the 5-km and 10-km aggregation. This problem occurs because the model tries to distribute data in zones lacking relevant land cover data. At the 10-km level, one 10-km zone was affected, and at the 5-km level, nine 5-km zones. The affected zones were located in coastal areas where the main part of the aggregation zone contained water. The problem was solved by relocating the holding data to the closest aggregation zone containing relevant land cover. This problem did not occur at the 1-km level, since the points were converted to a 1-km grid without any respect to the underlying land cover data. Nor was it a problem at parish level, as all farm records contain a parish-ID, and all parishes contained relevant land cover data. The parish distribution was however associated with another problem: Some of the holdings in the point data set were located outside the English parish data set. There are two reasons for this:

1) *The parish data set used in the re-distribution process does not cover all land.* When the parish polygon data set was converted to a 1-km grid, a priority was to maintain the original area (size) of the parish. Grid cells are limited in their shape, so some discrepancies occur at borders between polygons when they are converted to a grid. In order for the parish not to "grow" at the coastline, some of the land along the coast is not included, with a similar problem occurring at the border to Scotland and Wales.
2) *The methodology of allocation of holdings to the map.* When the original point data set of farm holdings was generated by Defra, holdings without any reliable geo-reference (10.3 % of the total number of holdings) were allocated a random geo-reference within the parish, with no consideration to the underlying land cover (M. Templeton, DEFRA, pers. comm., 2004). This means that the holdings can be allocated to land cover such as water, sand dunes etc. In addition to this, the parish map used in the holding allocation process is likely to be different from the parish data set used in this study.

The problem of holdings located outside the English parish data set was solved by allocating these points to the closest parish.

In the parish data set used in this study (from 1996), 11,121 English parishes are present with sizes within the range of 0.6 - 258 km^2 (Figure 9.5). As many as 269 parishes (2.4 %) only contained one 1 x 1 km grid square. When investigating grid squares with extreme emission values, the square containing the maximum livestock emission belonged to a 3 km^2 parish, and the square containing the maximum fertilizer emission belonged to a 2 km^2 parish. It may be concluded that small parishes are at larger risk of over- and underestimation of emissions. A suitable approach would be to aggregate the smallest parishes with neighbouring parishes to minimize the risk of these types of errors. It is however difficult to identify a size threshold for parishes to be aggregated, as this depends on the area of each holding, which in turn varies regionally (see Figure 9.1). An alternative approach to minimize the error is to aggregate the final emission map to a coarser resolution. This has been carried out for ammonia emission maps in the past where the final emission map was aggregated from a 1-km grid to 5-km grid resolution to reduce some of the uncertainties due to the MAUP (Dragosits *et al.*, 1998).

England Parishes

Number of parishes:	**11,121**
Average size:	**11.76 km²**
Minimum size:	**1 km²**
Maximum size:	**258 km²**

Parish size (km²)	Number of parishes
1-5	2,700
6-25	7,338
26-100	818
>100	25

1 km²	269
2 km²	349
3 km²	555
4 km²	846
5 km²	921

Figure 9.5. Statistics of the English parish data set showing various parish size groups, and number of parishes belonging to each size group.

Total ammonia emissions resulting from the 2-km non-disclosive data were different from the other emission maps, not only regarding the spatial location of emissions, but also regarding the magnitude of the total ammonia emission in England. Overall, total livestock emissions from the non-disclosive data were 5.4 % smaller compared with the emission calculation based on confidential data. This is because some rarer livestock categories, such as turkeys, horses and deer were not reported due to confidentiality issues and it was not possible to calculate emissions from these animals. For fertilizer emissions the emission difference was 0.46 % due to smaller areas of crops and grass reported for the same reasons.

9.3.3 Scanning data series

Figure 9.6 shows plots where the emission value for each 1-km grid cell at the different aggregation levels have been scanned for all datasets. When interpreting the scanning data series in Figure 9.6, it is important to note the scale and range of the y-axis. To facilitate comparison, the scale on the y-axis is the same for Figures b-e. The extreme values generated from the 1-km distribution could however not be presented at the same level of detail as for the other maps, and the scale on the y-axis is therefore different.

The scanning data series (Figure 9.6) show that the magnitude and number of extreme values are reduced with larger aggregation zones (due to the smoothing effect). The overall pattern of spatial distribution of extreme emission values is similar to the maximum emission values for the maps based on the confidential data situated in the same area (around raster-ID 77,000) for all levels. The variation between the scanning series is a result of both the scaling and the zonation effect. The scaling effect mainly impacts on the magnitude of extreme values, while variations in the spatial distribution of extreme values are explained by the zonation effect.

The scanning series of the emission data based on non-disclosive agricultural census data show a different spatial pattern than the maps based on the confidential data. Most of the emissions stay within 50 kg NH_3-N ha^{-1} and only very few grid cells stand out with extreme emission values. The most extreme emission values occur around raster-ID 90,000. The scanning series based on the confidential data also show an emission peak in the same area, but it is less distinct than in the non-disclosive data. The non-disclosive data appear to have 'lost' most of the peaks that the confidential data maintains (at a variety of magnitudes).

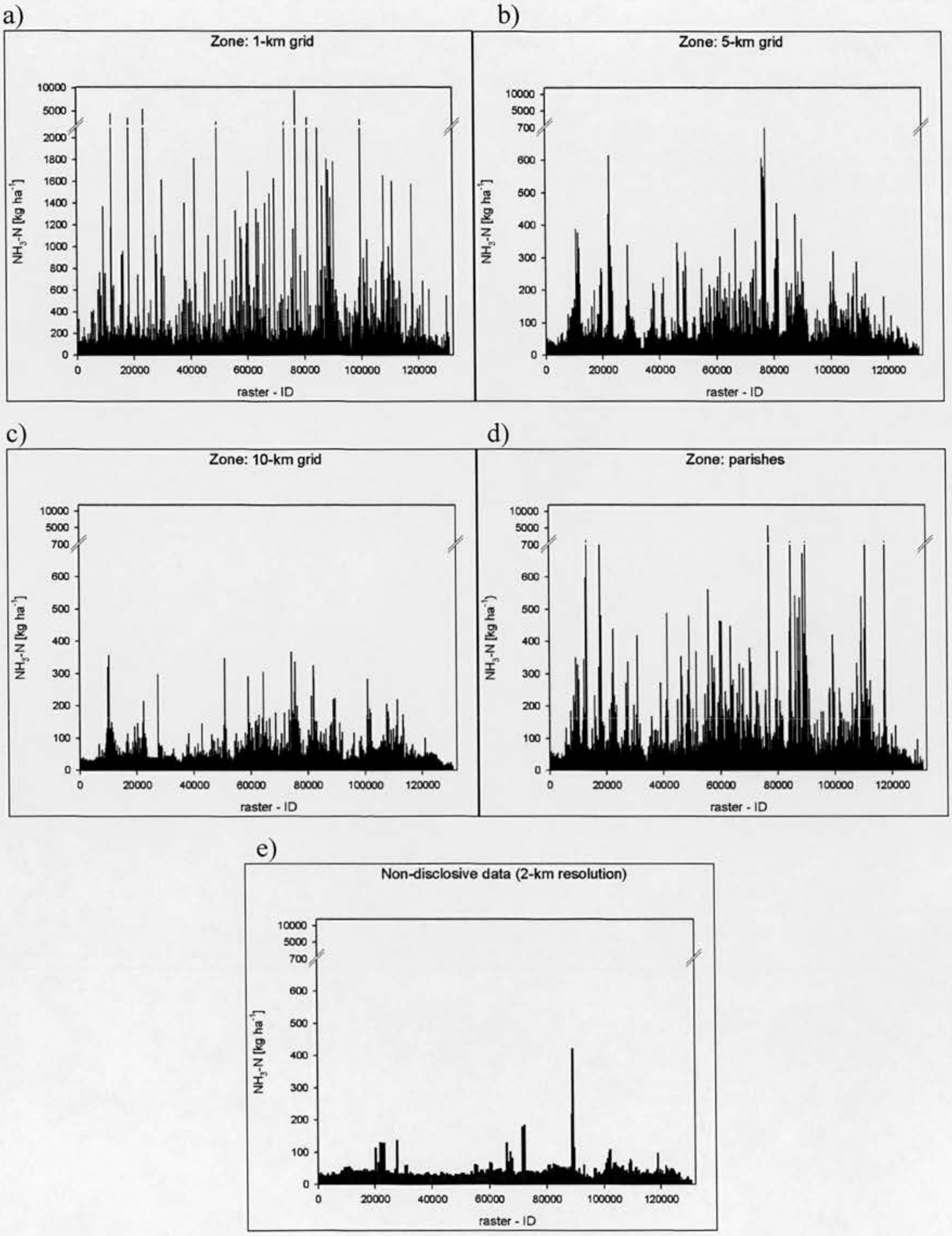

Figure 9.6. Ammonia emission values for each 1-km grid cell at different aggregation levels. Note that the scale on the y-axis for the 1-km level is different from the other scanning series.

9.3.4 Gridcells with unrealistic values representing the agricultural area

A problem with investigating the aggregation effects at different scales and zones is that there are no "true" values to compare with. Even the actual holding data cannot be used as a good reference, since in reality it is not likely that all farm land and livestock are located at the (point) location of the farm (see Section 3.8.1 and Figure 3.5 for details). It is common that the boundaries of the farm do not agree with the boundaries of the aggregation zone (e.g. parish), which complicates the assessment and quantification of the level of uncertainty.

One way to assess uncertainties further is to identify the number of 1 x 1 km grid cells containing more agricultural land according to the census data than the areal extent of the grid cell. Each 1 x 1 km square equals 100 ha, therefore it follows that the agricultural area in each square after the re-distribution process should be 100 ha or less. In many squares, however, the agricultural area (crop and grassland) is larger than the extent of the grid cell. Table 9.2 shows the percentage of agricultural squares at each aggregation level where the agricultural area exceeds 100 % and 110 %. Only 2.5 % of grid squares exceed the areal extent of the grid cell at the 10-km aggregation level, with 4.1% at the 5-km level and as many as 34.5 % at the 1-km level. In the parish distribution, 14.6 % of the squares had agricultural area overestimated. From these values it is clear that the smoothing effect of using larger aggregation zones reduces the number of overestimated squares as the aggregation zones increase in size.

Table 9.2. Total number of overestimated grid squares regarding crops/grass (area of crops & grass allocated > 100 and 110 ha).

Zone level	> 100 %	> 110 %
1-km grid	34.5 %	31.3 %
5-km grid	4.1 %	2.3 %
10-km grid	2.5 %	1.0 %
Parish	14.6 %	9.7 %

9.3.5 Comparison with land cover data

Another way to assess the accuracy of the agricultural census data is to compare them with another dataset representing similar information. In this study, the spatial distribution of agricultural land based on different zonal aggregations was investigated using the LCM2000 (Fuller *et al.*, 2002) dataset as a reference. This is the same land cover map that was used in the down-scaling from zones to 1-km grid in the AENEID model. The land cover data are not an exact representation of reality, and there are uncertainties associated with LCM2000 (see Section 3.8.2 for details), but it may give an indication of the areal extent of agricultural land in each 1-km grid cell. The comparison is limited to comparing agricultural land, i.e. crop and grass area, and is not suitable for the livestock categories in the census. Uncertainties for the livestock categories are more difficult to estimate, firstly because there are no other datasets of animal distribution available for comparison, and secondly, because the livestock are not distributed as such, but rather as ammonia sources.

In the LCM2000 dataset, the land cover classes arable cereals, horticulture and non-rotational agriculture, were aggregated to represent arable land (crops and horticulture) while grassland was assumed to be represented by improved grassland. The total area of arable land in LCM2000 is thus 48,258 km^2 compared with 44,593 km^2 reported in the agricultural census. The agricultural census data therefore reports 7.6 % less arable land than LCM2000. The area extent of grassland reported on the other hand is 15.6 % higher, with the total area of improved grassland in LCM2000 being 30,221 km^2 compared with 34,921 km^2 in the agricultural census.

A limitation of comparing grassland and crop areas in the two data sets is that both datasets are uncertain. Although they represent similar types of data, the data collection methodologies are very different. Agricultural land in LCM2000 has been defined by interpreting and classifying satellite images, while the agricultural census is based on information given by the farmers. Interpreting and, in particular, differentiating between grass and crops in the satellite images for land cover classification is not an entirely straightforward process. Uncertainties in the satellite land cover classification can result from short-term grass being confused with crops, especially for large areas of dry grass, or where there is a short rotation of grass/arable (Fuller *et al.*, 2003). In addition, it is likely that different criteria were applied to allocate agricultural land to arable or grassland, i.e. farmers may include grassland other than what is defined as "improved grass" in LCM2000. The difference in the area of grassland between the two datasets is therefore probably due to different criteria for classifying grassland in LCM2000 and in the agricultural census. In this study, the area of crop land and grassland at all aggregation levels was compared with LCM2000. Three different methods were applied to compare LCM2000 with the re-distributed zonal maps: *difference maps, scattergrams* and *an acceptability criterion.*

Difference maps

Difference maps based on aggregated parish data were calculated for crops and grass by subtracting the agricultural census data from LCM2000, to highlight and identify discrepancies between the two datasets (Figure 9.7). The difference maps show areas where the agricultural census parishes contain either substantially larger or smaller areas of crops and grassland per km^2 than the LCM2000 data aggregated by parishes. Minor differences (<10 %) are shown as a separate category.

Figure 9.7.a shows a reasonable correspondence between the two datasets regarding arable land in the grassland dominated area of NW England, as well as in urban areas such as London. In other parts of the country, the difference map shows a very speckled appearance, characterised by parishes with higher and lower values in close proximity. In these areas it is likely that the surplus of land within one parish surrounded by parishes with "shortages" of land should in fact be distributed among these parishes. This may be explained by the fact that, although a holding is registered in one parish, its land may actually be situated in another parish.

The two datasets were also quantitatively compared at parish level for parishes that contained arable or grassland, respectively, in both LCM2000 and the agricultural census statistics. The average difference for all parishes was 50 % for arable and 121 % for grassland, with a Root Mean Square Difference (RMSD) of 83 % for arable and 2,637 % for grassland. This comparison suggests that the grassland areas in the two datasets do not correlate very well at parish level.

Figure 9.7.b shows that the difference in the grassland data between the agricultural census and LCM2000 (15.6 %) is more consistent across the country than for arable land. Arable dominated areas (such as eastern England) as well as urban areas show a good agreement between the two datasets, while grassland dominated areas (e.g. NW England) have less grassland in the agricultural census data than in LCM2000.

In SW England there appears to be some correspondence between the surplus of arable land (Figure 9.7.a) in the agricultural census and the shortage of grassland (Figure 9.7.b). This suggests that either there is a discrepancy in the definition of the same class in the two datasets or that there may be some concern over the accuracy of the satellite classification in this area. This could be due to grass being misclassified as arable, or due to short rotations in areas that are grass at certain times of the year and arable at others, and so uncertainties in the satellite data may be important.

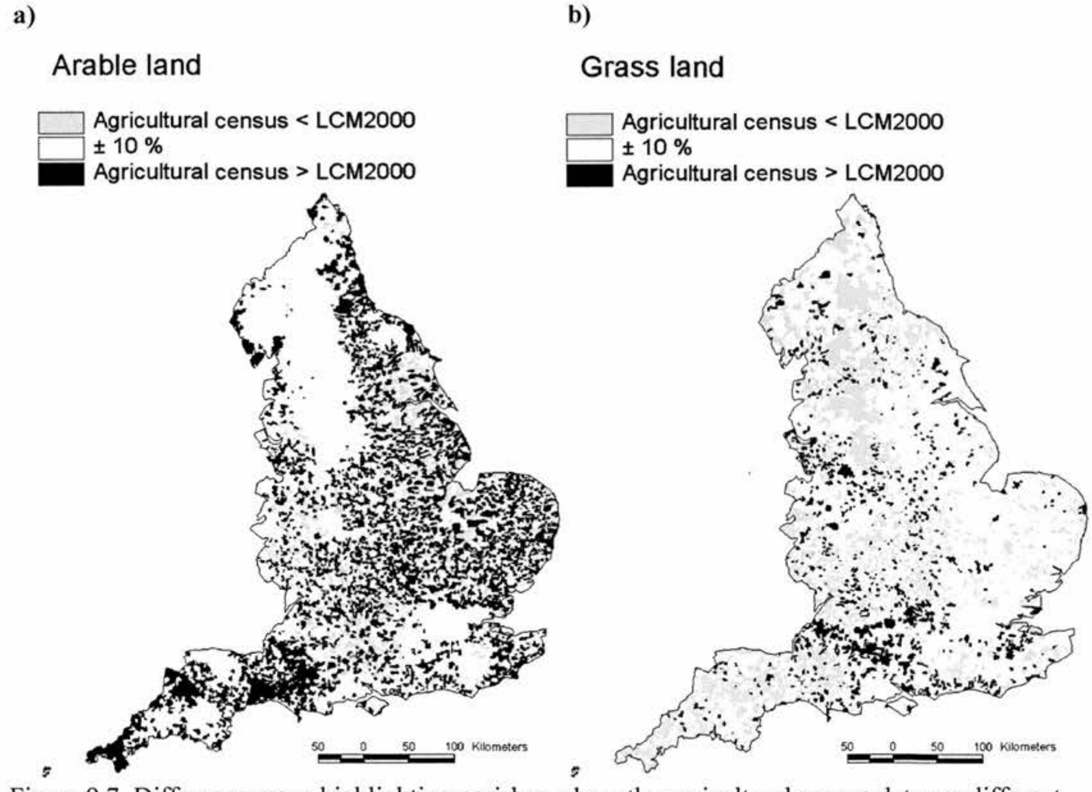

Figure 9.7. Difference maps highlighting parishes where the agricultural census data are different from LCM2000 regarding a) arable land b) grassland.

Scattergrams

Figure 9.8 shows scattergrams comparing arable land in LCM2000 with agricultural census data for all four zonal systems (1-km, 5-km, 10-km zones and parishes). Table 9.3 shows the coefficients of correlation (R^2) derived for all four aggregation levels. Scattergrams and R^2 values were not calculated for the non-disclosive (2-km grid) data, as they are not directly comparable with LCM2000, due to their different origin and spatial resolution.

The smoothing effect was again clearly exemplified by both the scattergrams and the coefficient of correlation. R^2 increases with zone size, showing that values are more statistically stable for larger aggregation zones.

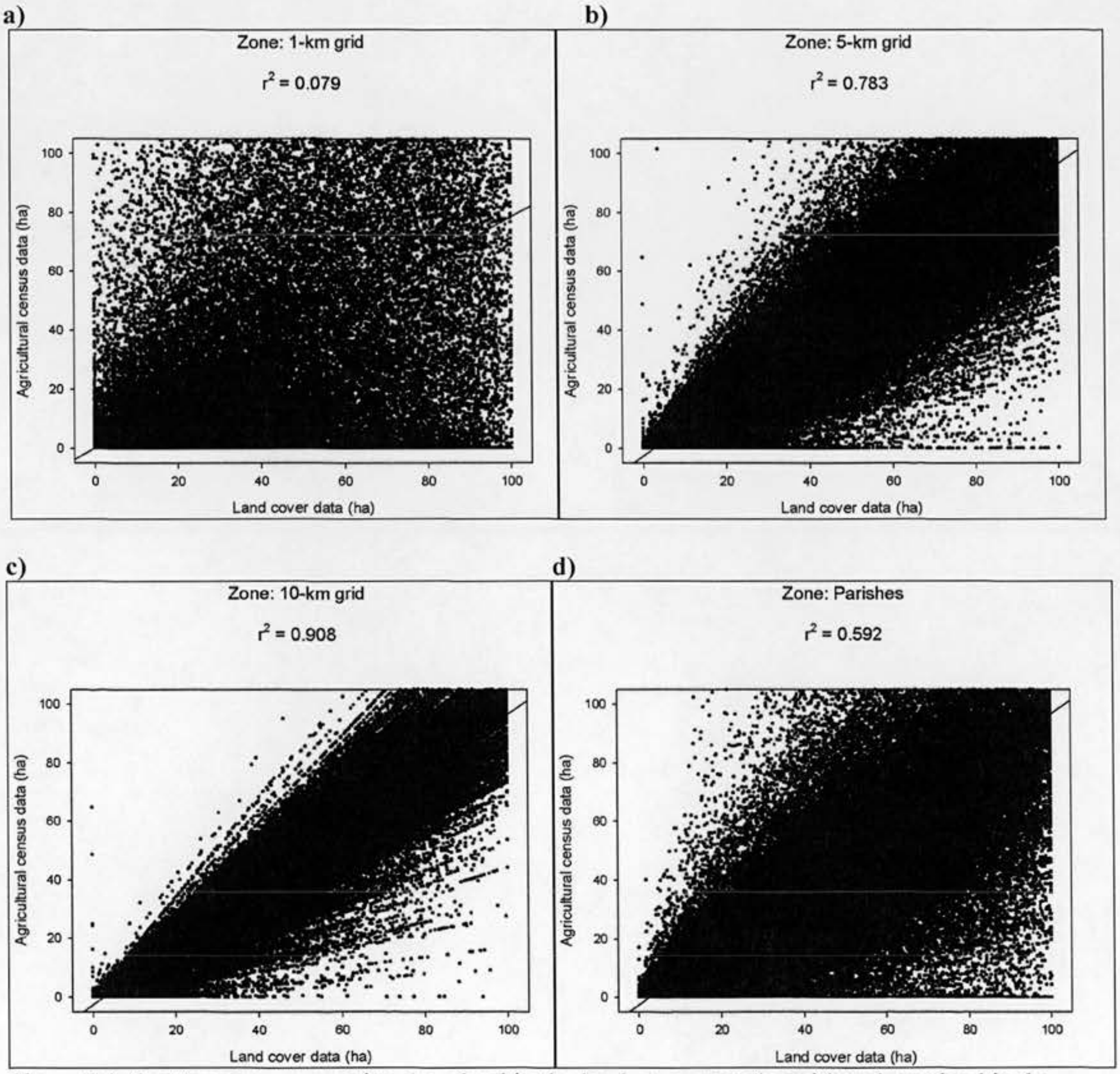

Figure 9.8. Scattergrams comparing crop-land in the land cover map (x-axis) and cropland in the agricultural census data (y-axis) at different aggregation levels.

Table 9.3. R^2-values when comparing crop and grassland in the LCM2000 with the agricultural census data distributed at different aggregation levels.

Zone	R^2 - cropland	Regression values (cropland)	R^2 - grassland	Regression values (grassland)
1-km grid	0.079	Y = 0.16 + 0.78X	0.127	Y = 6.00 + 0.89X
5-km grid	0.783	Y = -2.59 + 0.99X	0.692	Y = 1.27 + 1.09X
10-km grid	0.908	Y = -2.50 + 0.99X	0.834	Y = 0.62 + 1.12X
Parishes	0.592	Y = -2.49 + 0.99X	0.639	Y = 1.36 + 1.10X

Acceptability criterion

The result of comparing LCM2000 with the four distribution maps regarding the acceptability criteria of ± 25 % (± 25 ha) for arable and grassland is shown in Table 9.4 and Table 9.5. The tables clearly show the smoothing effect, i.e. the increasing number of grid cells falling within the acceptable range as the size of the aggregation zone increases. Only about 51 % of the cropland and 67 % of the grassland meet the acceptability criterion at the 1-km level. The number of acceptable squares increases steadily with zone size, with 90 % for crops, and 94 % for grass at the 5-km level, and 97 % for crops and 96 % for grass at the 10-km level. The number of squares within the acceptable range for the parish distribution is 82 % for crops and 91 % for grass.

Table 9.4. Comparison of cropland in LCM2000 with the agricultural census data at different aggregation levels. (Acceptability criteria: ± 25 ha / 1 x 1 km grid cell, i.e. 25 %)

Zone level	acceptable squares (+/- 25%)
1-km grid	50.7%
5-km grid	89.5%
10-km grid	96.9%
Parish	82.4%

Table 9.5. Comparison of grassland in LCM2000 with the agricultural census data at different aggregation levels. (Acceptability criteria: ± 25 ha / 1 x 1 km grid cell, i.e. 25%)

Zone level	acceptable squares (+/- 25 %)
1-km grid	66.7%
5-km grid	93.6%
10-km grid	96.0%
Parish	90.8%

9.3.6 Evaluating the results

A range of different methods, which all have different strengths and weaknesses, was applied in this study to evaluate the effect of the Modifiable Areal Unit Problem. Many of the methods have clearly shown the smoothing effect that occurs with increased aggregation zones. This smoothing effect has the advantage of making the result more statistically stable, but a disadvantage is that spatial detail is lost. If statistically stable values, e.g. increased R^2 values and the closest resemblance to LCM2000 was the sole criterion for optimal zonal scale, then clearly the larger the zones, the better the result.

It is difficult to conclude what zone size is preferable for the data in this study, if indeed there is a single value, as it may differ regionally. The parish data contain a large range of zonal sizes, with small parishes containing sometimes too much spatial detail, while larger parishes are more statistically stable. Some of the extreme and unrealistic values resulting from a small parish size can be remedied by aggregating the smallest parishes with a neighbouring parish. Extreme values as a result of a limited amount of relevant land cover data within a parish are more difficult to pinpoint and remedy.

For agricultural census data as input to spatial emission models, parish data may still be the best option. There is a danger in using a zonal system that is different from the parish zones, as some of the holdings (with no reliable geo-reference) are placed randomly within the parish, and not according to their exact location. When using the parish data, at least the farms are allocated to the correct parish. In addition to this, the choice of zonal system is limited by data availability, and for some parts of the UK, parish-based agricultural statistics are the only option available.

9.3.7 Ways to reduce uncertainties due to the MAUP

Many ways to reduce uncertainties due to the MAUP have been suggested in literature, but few practical solutions to the problem exist. One option is simply to estimate uncertainties through sensitivity analysis by evaluating which variables are sensitive to variations in scale and zoning effects and how sensitive they are (Jelinski and Wu, 1996).

Sadahiro (2000) suggests using areal interpolation methods as a tool to yield more accurate estimates, e.g. methods using supplementary data to improve the accuracy of zonal data estimates. An intelligent interpolation method has already been applied in the context of modelling ammonia emissions by incorporating land cover data in the modelling process of AENEID (Dragosits *et al.*, 1998), see Section 3.5 for further details.

Openshaw strongly advocates '*an optimal zoning approach*' as a solution to the MAUP (Openshaw, 1977; Openshaw and Rao, 1995; Openshaw and Alvanides, 1999; Alvanides and Openshaw, 2002). Openshaw suggests that uncertainties could be reduced if the zoning system was based on the characteristics of the data, e.g. farm density, rather than artificial (e.g. parish) boundaries. Alvanides and Openshaw (2002) suggest that zone design is *'a scientific way of manipulating aggregation effects in order to produce optimal zoning systems for specific purposes'*. The problem with using parishes as aggregation zones is that they are constructed with regard to administrative borders (artificial units) rather than natural units. Parishes vary in size, shape, number of farms and type of data. When dissimilar units are aggregated, the loss of information is higher than if the areal units are similar to begin with (Crawford and Young, 2004). Zone design may be a powerful tool when small zone entities, such as the holding data, are available and when scale and aggregation method for these small entities can be adapted to the purpose of the study.

The zoning systems investigated in this study were designed without any consideration of the characteristics of the data. The grid zones of 1-km, 5-km and 10-km resolution were chosen due to their regularity and the fourth zonal system, parish zones, were chosen as the agricultural census data for England have been distributed at parish level in the past. It is likely that a better spatial emission inventory would be obtained if the zone design was based more on the characteristics of the data. A point-to-polygon approach (Thiessen polygons), where the farm statistics are distributed within these polygons rather than within the parish, and where the zone design is adapted to the farm density, could be a suitable solution, hence reducing uncertainties due to the MAUP. However, this approach is only applicable when holding data are available as individual points, and, in the case of agricultural census data, the best available level is normally a parish- or grid- based distribution.

Sadahiro (2000) suggests that the problem of the MAUP may be reduced if the aggregation zones are smaller than the target zone. This method presupposes that the final emission resolution (the target zone) is larger than the aggregated zones. In a way, this has been applied in the past when the final ammonia emission map was aggregated to a 5-km resolution (Dragosits, 1999). The actual modelling in AENEID is carried out at a 1-km grid by down-scaling the aggregated zonal data from parish level to the 1-km level. Although the aggregation zones were sometimes larger than the target zones, the modelling resolution (1-km grid) was not. The 1-km grid was then upscaled to the coarser final resolution of 5-km. The extent of the problem of the MAUP on spatio-temporal emission inventories (monthly emission maps) is not considered to change with season, as the same aggregation zones and distribution methodology is applied for all months.

It may be tempting to keep the 1-km fine modelling resolution for the final emission map. Due to the problems of the MAUP exemplified in this chapter, however, it is recommended that some level of up-scaling is carried out in the final emission map. If small zones (i.e. parishes < 5 km^2) are aggregated with neighbouring parishes, a final grid resolution of 2 x 2 km might be acceptable.

If the resulting NH_3 emission map is to be used as input to e.g. an atmospheric transport model, the resolution could be maintained at 1 km resolution, if the modeller using the data is aware of the associated spatial uncertainties. Modelling the atmospheric dispersion and deposition at a 1 km level should generate more accurate results than using e.g. 5-km resolution emission maps. It can be argued that atmospheric dispersion modelling will introduce an additional element of smoothing.

9.3.8 Effects of the MAUP on spatial ammonia emission inventories

This study has shown that the MAUP can have a significant effect on the location of emissions in spatial inventories where aggregated zonal data are used. If the aggregation zones are small, there is a substantial risk of the MAUP intensifying emissions which results in unrealistic peaks. These emission peaks will inevitably be propagated in atmospheric dispersion modelling of NH_3, thereby overestimating the ammonia concentration and deposition in some areas while underestimating them in other areas. These propagated errors may consequently falsely highlight areas that exceed the critical load of nutrient N. If the aggregation zones are, on the other hand, too large, true emission peaks are likely to be smoothed out, hence giving the impression of fewer areas exceeding the critical load than should be the case. Spatial accuracy and decreasing spatial uncertainties are therefore important tasks. It is recognised that it is impossible to totally eliminate the effects of the MAUP, but raising the awareness of this type of problem, especially among people working with zonal data as input to spatial emission inventories, may lead to a better understanding and interpretation of the results.

The problem of the MAUP is also relevant for other spatial emission inventories or any other analysis where zone-based data are used. Uncertainties due to the MAUP can be reduced by aggregating very small aggregation zones with neighbouring zones, using supplementary data to distribute the statistics within the zone (areal interpolation methods), or by aggregating the final result to a coarser resolution.

9.4 Conclusion

In this study, England was used as a test area to study the effects of the MAUP. Agricultural census data at farm level (point data) were obtained and amalgamated to different zone levels: 1-km, 5-km, 10-km and parish level, before they were used in the AENEID model to estimate the spatial distribution of NH_3 emissions. The results of using the census data at different levels of amalgamation were compared as a means to estimate the effects of the MAUP on the emission inventory. A data set containing already spatially distributed agricultural census data at 2-km resolution using a different methodology, was also included in the study. These data had been aggregated to a much greater level to preserve confidentiality of farmers at the level of single census items.

Ammonia emissions are directly linked to the spatial location of agricultural sources. Finding the most likely location of these sources is therefore essential for spatial ammonia emission inventories. This study has shown that the size and location of aggregation zones for the agricultural census statistics strongly affect the location of the emission results. If the zones are too small, this may result in false emission "hot spots", i.e. extreme emission values that are not actually at that location. Conversely, if the zones are too large, detail may be lost, and the smoothing effect may give a false impression of the magnitude of emissions in some areas of the country. The results of the study indicate that the MAUP has a significant effect on the location of emissions in spatial inventories, where amalgamated zonal data are used.

The findings in this study are not only relevant for spatial ammonia emission inventories, but can be applied in other spatial emission inventories or any analysis where zone-based data are used. When dealing with aggregated point data, the MAUP will always be present, unless the spatial distribution is totally homogenous, which is unlikely for most distributions. Homogeneity is certainly not common in parish aggregated agricultural census data, as they are based on artificial administrative boundaries. Furthermore, uncertainties due to the MAUP varies within the country, as the aggregation zones (parishes) tend to be larger in Scotland and Wales (where parish groups are used), compared with England. The spatial distribution error can however be reduced by aggregating extremely small parishes with neighbouring parishes, by incorporating supplementary data (e.g. land cover data) to distribute the census items within each zone and by aggregating the final result to a coarser resolution.

10 Discussion and conclusions

10.1 Introduction

The Institute of Geography, University of Edinburgh, has together with the Centre for Ecology and Hydrology (CEH), been working on modelling the distribution of ammonia emissions since 1992 e.g. (Eager, 1992; Dragosits *et al.*, 1996; Dragosits *et al.*, 1998; Dragosits, 1999; Sutton *et al.*, 2000; Dragosits *et al.*, 2002). This thesis fits into this wider programme of work on ammonia, which also covers consideration of the mechanisms of emission and deposition, development of process based models and consideration of novel strategies for ammonia abatement.

The aim of this thesis was to improve the original AENEID model (Atmospheric Emissions for National Environmental Impacts Determination) to better reflect the geographical distribution of ammonia emissions on a UK scale. The ammonia emissions are modelled at a 1-km resolution, and generalised to 5-km resolution for use in the National Atmospheric Emission Inventory (NAEI) and in atmospheric dispersion models, as well as for environmental risk assessment. Limitations in the original AENEID model such as the inability to handle regionally and temporally varying emission potentials, and the unrealistic emission peaks resulting from the modelling of pig and poultry farming had been identified as major uncertainties, and these areas constituted the main focus for this thesis. The original AENEID model was modified and improved to reduce some of the uncertainties associated with these areas. Furthermore, the spatial extent of the model was expanded to include the Isle of Man, and updated input data (a new landcover map and updated emission potentials) were incorporated in the new emission estimate. This chapter summarises the work described in this thesis, and discusses the new modelling approach to distribute ammonia emissions in the UK.

10.2 Development of the new AENEID model to disaggregate spatially the ammonia emission inventory

10.2.1 Objectives and background

The main objective of this thesis was to improve the original methodology of the AENEID model, to reduce uncertainties in the spatial emission inventory and to provide more robust spatial emission estimates of ammonia in the UK. Ammonia emissions contribute to the eutrophication of N sensitive ecosystems and acidification of soils and water bodies. The spatial variability of NH_3 sources in the country, together with the fact that NH_3 is highly reactive, and therefore deposits close to the sources, highlight the importance of providing reliable spatial emission maps of ammonia. Locating the emissions accurately is essential, because AENEID model output is used to assess potential environmental impacts, such as exceedance of deposition thresholds (critical loads). Incorrect mapping of ammonia emissions, therefore leads to errors in assessing potential environmental impacts.

10.2.2 The new AENEID methodology to disaggregate ammonia emissions in the UK

The new modelling approach developed in this thesis is based on the original AENEID model (Dragosits, 1999). The basic principles of the original model were maintained, but the model was modified to be more flexible and to reduce some of the major uncertainties associated with NH_3 emission modelling. The programming structure (FORTRAN) was modified to be able to handle the input variables (apportioning percentages) either as input grids, or in a parameter file, rather than the hard coded values in the original programme. This allows for more flexibility of the model, as it is able to handle temporally resolved (monthly) inputs, as well as spatially variable (grid based) input for application in e.g. the coupled NARSES-AENEID model, where outputs from the NARSES N-flow module are used as input to AENEID. The AENEID model takes less than 15 minutes to run, but with the new poultry sub-model, the run-time is extended to about 2 hours.

A major uncertainty identified in the original AENEID model was the presence of very large emission peaks resulting from distributing all emissions from pigs and poultry in the aggregation zone of origin. A sub-model for the distribution of pig and poultry manure for large farms was therefore developed, with a new re-distribution approach based on maximum application rates of manure over a larger area, independent of the size and shape of the aggregation zone from which the manure originates. The submodel also takes account of manure output from other livestock, and this results in a more realistic distribution of NH_3 emissions.

Another important uncertainty identified in the original AENEID model refers to the temporal resolution, as only annual emission maps were produced in the past, which did not address the strong seasonal patterns of ammonia emissions. Monthly emission maps were calculated in this thesis, incorporating seasonal variability in agricultural practice. One of the most important temporal uncertainties, which also varies substantially between different parts of the UK, is the length of the grazing season, as the emission potential of cattle depends on whether they are grazing or kept indoors. This uncertainty was assessed by modelling the length of the cattle grazing season in the UK, and adjusting the emission potentials of dairy cattle accordingly.

The emission potentials applied in the AENEID model are based on the Unit Approach, i.e. the emission potentials are expressed as mass NH_3 emitted per unit of activity (such as grazing, housing, manure storage and landspreading). However, moving towards a Mass-Flow approach (where the emission potentials are expressed as a percentage value of the previous emission stage) allows for more flexibility to the model, as spatial variability in emission potentials can be incorporated, which is more suitable for estimating local variations in NH_3 emission potentials. By linking the AENEID model to the NARSES N-flow module (a process based NH_3 emission model, based on the Mass Flow approach), a coupled approach was developed, to allow for the application of variable emission potentials in the AENEID model.

10.3 Results of the new AENEID model to calculate ammonia emissions for the UK

10.3.1 Ammonia emission maps

The spatial NH_3 emission inventory was updated to 2000, applying the new AENEID approach for the whole of UK, including the Isle of Man. The new ammonia emission map (Figure 8.6) showed that non-agricultural emissions are dominant in urban areas, while agricultural emissions are the dominant NH_3 sources in most rural areas. Lowland areas are dominated by emissions from livestock, particularly cattle emissions, while emissions from sheep are dominant in upland and hill areas. Emissions from pigs and poultry are more localised. Areas with intensive agricultural activity dominated by livestock are mainly situated in Wales, the western part of England and in south-eastern parts of Scotland. Areas dominated by NH_3 emissions from fertilizers and crops, are situated mainly in eastern parts of England.

Results for year 1996 based on the new AENEID approach were analysed and compared with the 1996 emission estimates based on the original AENEID model, by Dragosits (1999). The comparison suggests that the new methodology provides a smoother emission pattern with less extreme emission peaks, mainly as a result of the pig and poultry sub-model incorporated in the new model, which distributes manure spreading emissions from pigs and poultry across a greater area. Improvements in the emission potentials applied have also led to overall lower emission estimates in the new AENEID emission inventory.

10.3.2 Seasonal variations in ammonia emissions in the UK

Monthly NH_3 emission maps were calculated by incorporating monthly emission potentials, and spatial apportioning percentages were adjusted to seasonal variations in farming practice. These maps were able to capture the general seasonal trends in ammonia emission that occur during a year, as the modelled temporal emission trend corresponded fairly well with the temporal trend in NH_3 concentrations (Section 5.7.6). The monthly calculated NH_3 emission result showed a strong seasonal emission pattern, with the highest emissions during springtime (March and April) and the lowest emissions during summer (May to July). This emission pattern was mainly influenced by cattle emissions, which contribute 54 % of the total agricultural ammonia emission in the UK. Ammonia emissions from cattle vary considerably throughout the year as a consequence of distinct housing and grazing seasons, with significantly lower emissions during the grazing season than when cattle are housed. Another important temporal aspect affecting the seasonal pattern of NH_3 emissions is the application of manures and fertilizers that occurs mainly in springtime, resulting in higher emissions compared with the rest of the year.

The monthly NH_3 model took into account both the temporal variation in the magnitude of the ammonia emission potential and the spatial variation of those temporal changes, i.e. the change in location of emission sources in the landscape during the year. Such intra-annual variation in NH_3 emissions needs to be considered in addition to the calculation of annual NH_3 emission estimates, to target abatement strategies efficiently throughout the year. Furthermore, annual estimates fail to capture seasonal peaks in emissions that are important in relation to exceedance of NH_3 critical levels.

10.3.3 Temporal changes in ammonia emissions during 1990 to 2000

Spatial emission inventories for 1990, 1996 and 2000 were calculated and analysed to investigate trends in the emission pattern during the 10-year period. The results suggest that a steady decline in NH_3 emissions has occurred during the period, which is mainly due to a reduced number of livestock and lower emission potentials for livestock due to changes in management practice. During the period there was a gradual decline in fertilizer application and a smaller proportion of this was applied as urea. Overall, the agricultural NH_3 emission decreased by 16 % between 1990 and 2000.

10.3.4 Evaluation of the AENEID model

The emission results of the new AENEID model for year 2000 were evaluated with the FRAME model (Section 8.5) and independent measurements from the National Ammonia Monitoring Network, NAMN (Section 1.5.4). The comparison suggested a fairly good fit of the magnitude and spatial variability of ammonia concentrations at a national scale, with an R^2 value of 0.6. The evaluation further suggests that emissions are overestimated by AENEID in cattle, pig and poultry dominated areas compared with the measurements in the monitoring network, while sheep and non-agricultural areas show a good correspondance (Section 8.5).

10.4 Uncertainties in the UK ammonia emissions inventory

Modelling NH_3 emissions involves using different input data, some of which are spatially distributed, to calculate NH_3 emissions using mathematical expressions. Uncertainties in the modelling result are therefore associated with both the quality of the input data and the mathematical expressions applied in the model. The uncertainties may relate to the magnitude of emissions and/or the spatial location of emissions, which are further influenced by temporal uncertainties due to the seasonal variability in environmental conditions and farming practice that occur within and between years.

Uncertainties in the AENEID model are associated with the data representing the sources (i.e. the agricultural census statistics), the datasets used in the modelling process (landcover data, source strength and agricultural practice), and the modelling methodology itself. The most important input data for the model are the annual agricultural census statistics. Uncertainties in the census data may either be due to statistical errors or associated with the location of the farms. Census data are collected at farm level, but aggregated to a coarser spatial resolution in order to maintain confidentiality. There is little relationship between farm boundaries and aggregation zone boundaries, i.e. a farm may only partially be located within the aggregation zone. Farmland and associated NH_3 emissions in each aggregation zone can therefore be over- or underestimated by varying degrees, especially in small aggregation zones. Uncertainties due to spatial aggregation of data in general are commonly referred to as the Modifiable Areal Unit Problem (MAUP). In this study, the manifestation of the MAUP is that the emission result changes depending on the type of aggregation zones applied. The MAUP was assessed in this study by calculating emission maps based on different sets of aggregation zones. The results suggest that the MAUP has a significant effect on the location of emissions in spatial inventories when aggregated data are used. The desire for increased resolution in order to identify localised impacts must be tempered by the need to aggregate to a coarser resolution in order to minimise the likely errors introduced by the MAUP.

The AENEID approach distributes the agricultural census data as NH_3 sources in the landscape within the aggregation zone, onto the landcover type where the emissions are most likely to occur. Uncertainties in the land cover dataset applied in the re-distribution process, and the assumptions applied to allocate emissions to landcover types therefore influence the spatial location of emissions. Currently, UK average farming practice is assumed in the AENEID model in relation to the spatial re-distribution of NH_3 sources to landcover types in each aggregation zone. Any regional variations in these relationships are not taken into account, but uncertainties could be reduced if management data for individual farms or maps of regional practice differences were available.

The assumption that all emissions occur in the aggregation zone of origin is on average acceptable for land-based farming of cattle and sheep, but for emissions from intensive pig and poultry farms, where excess manure is often transported over long distances, the spatial uncertainties are much greater. A pig and poultry sub-model was therefore developed and incorporated into the new AENEID model, which breaks the link between NH_3 emissions and the aggregation zone. The methodology is based on the manure capacity of the area surrounding large farms and emissions from manure spreading are distributed over a much larger area. This sub-model was based on a detailed study of poultry facilities in Scotland, and was considered to provide more realistic emission outputs.

Uncertainties in the emission potentials applied may be reduced if the emission potentials are estimated based on local conditions and practices. The AENEID model was therefore linked to the process-based NARSES N flow-module, a model that accounts for some regional variations in emission source strength, to assess the possibility to use variable emission source strength data in the AENEID model. The initial results of linking the two models are promising, but further work is needed in the way output from the NARSES model is provided, to assure a more streamlined data transfer.

Emission inventories generally tend to be calculated on an annual basis. Annual emission inventories fail to capture temporal variations due to meteorological conditions and seasonal trends in agricultural practice, and are therefore associated with temporal uncertainties, both regarding the magnitude and spatial location of emissions. When temporal variability in farming practice was introduced and incorporated into the AENEID model, a strong seasonal emission pattern was evident, highlighting the importance to capture seasonal NH_3 emission trends.

A major uncertainty in the applied emission potentials is the spatial variability of the cattle grazing season in the UK. The potential length of the grazing season was therefore estimated, and the modelling results suggest that the potential cattle grazing season varies significantly within the UK, and also between years. Grazing animals are associated with significantly lower overall emission rates per unit of time than housed cattle and this is likely to have a significant impact on spatially distributed NH_3 emissions.

10.5 Levels of spatial scales in the ammonia inventory

Ammonia emissions are characterised by a very large variability at a local scale, and it is therefore crucial to apply a suitable spatial resolution when mapping ammonia emissions. When determining which scale to apply in a model, the purpose and outputs of the model should be considered, as well as the type and scale of available input data applied in the model. Currently, the AENEID model calculates ammonia emission maps at 1-km resolution (the processing level), but these maps are aggregated to 5-km resolution (the distribution level) for application in atmospheric transport models. It may be argued that the processing level should be carried through to the atmospheric transport models to further improve the result, as the variability within each 5-km grid cell may be considerable. If a coarse spatial resolution is applied, information about emission peaks or low emissions occurring at a more local level will be lost. Applying these emission values in an atmospheric transport model and subsequent assessment of critical loads exceedances, may underestimate exceedances in semi-natural areas in close proximity to intensive agricultural areas.

A disadvantage of applying the results at 1-km level is that the spatial uncertainties associated with emission estimates at 1-km grid resolution are likely to be greater than at 5-km level. The result of the MAUP study suggested that large spatial uncertainties occur at the 1-km level, but aggregating the data to a coarser spatial resolution reduces some of these uncertainties. Another disadvantage is that the 1-km resolution may conflict with confidentiality constraints imposed on the agricultural census data. If 1 x 1 km outputs from disclosive inputs are considered to be disclosive, then the model calculation would have to be based on non-disclosive data input, which would further increase the uncertainty in the emission result. However, the basic principles of the AENEID ammonia modelling approach ensures that confidentiality of the data input is maintained by linking emissions to land cover within the aggregation zones, so that the emission for each component at a 1 x 1 km level cannot be directly related to numbers of animals from each given farm. Furthermore, only reporting the output ammonia emissions in aggregate form by combining different source stages (housing, storage, grazing, manure spreading) and different source types (e.g. cattle, sheep, pig and poultry) into three categories of livestock, fertilizer/crop and non-agricultural, helps to maintain confidentiality of the input data. Through a combination of these operations, disclosive agricultural census data can be used as input to the atmospheric dispersion model, which result in non-disclosive output.

Another disadvantage of applying emission estimates at 1-km resolution in atmospheric transport models is that the data volume would increase 25-fold. While this has been a major hurdle when modelling at a national scale in the past, recent development in computing technology have overcome this, and the increased level of detail would be valuable for regional or local studies. The main advantage of calculating 1-km inventories is that this allows for a more detailed analysis of the results, which in turn improves model testing, and validation of the model.

10.6 Recommendations and transferability issues

This study showed that the result of the ammonia emission inventory is very sensitive to the agricultural census data used as input data to the model, both regarding the magnitude of emissions (statistical uncertainties in the census), and spatial uncertainties (how the agricultural statistics are aggregated). To provide robust spatial emission inventories, it is therefore important to use agricultural statistics at the best possible level of detail, i.e. data that have not undergone any form of data modification to maintain confidentiality. Access to agricultural census data at farm (holding) level provides the opportunity to handle the data in the most flexible way, i.e. the data can be aggregated at a resolution suitable for the application, for instance through a point-to-polygon method, to reduce uncertainties in the modelling result.

Furthermore, there is a need for consistency with categorisations of livestock between different administrative areas in the UK, which would facilitate the processing of agricultural census data as input to AENEID. It is also recommended that the different devolved regions responsible for collecting the agricultural census data aim at reducing spatial and statistical uncertainties in the agricultural statistics further. This is particularly important for large intensive pig and poultry farms, which are sometimes reported to be located at the location of the head office rather than at the exact location of the farm operation.

When the original AENEID model was run in the 1990s to re-distribute ammonia emission sources in the landscape, it took several hours to run the model. Today, despite the fact that a larger number of emission sources (animal and crop categories) are used, mainly due to advances in computer technology, the model only takes about 10 minutes to run. However, after the implementation of the iterative poultry model, the run-time has increased to about two hours. However, the preparation of the datasets used as input to the model can be time-consuming. The agricultural census data need to be aggregated into the 46 input categories. This process would be straightforward in a country where the agricultural statistics are collected and distributed by one organisation. However, as the data had been collected by different organisations in the devolved regions in the UK, preparing the agricultural census data for input to AENEID is more complex.

Another time-consuming task, before the AENEID model can be run, is the derivation of the emission potentials and apportioning constants used as inputs to the model. These model input data are based on the national ammonia emission inventory (IAEUK) and require processing to derive at the ammonia emission for each individual manure management stage (grazing, housing, manure storage and manure spreading) for each livestock category. Providing that this type of emission inventory is available, and that other necessary input data (agricultural statistics and landcover data) are available, the AENEID model could be modified to be applied for application in other countries.

The AENEID model can potentially also be used to calculate spatial emission inventories for other emissions, providing that aggregated emission source statistics can be linked to landcover types and therefore re-distributed within the landscape. For instance, the AENEID methodology has been applied to distribute methane emissions (Sutton *et al.*, 2004a; Sutton *et al.*, 2006).

The AENEID model calculates NH_3 emissions at a 1-km grid resolution, but the final emission result is aggregated to a 5-km resolution to reduce some of the spatial uncertainties due to the MAUP, and as part of the disclosivity agreement for access to the agricultural census data. If future emission maps are to be presented at a higher resolution, it is recommended that some level of up-scaling in the final emission map is carried out. If the agricultural statistics in the smallest parishes (i.e. parishes < 5 km^2) are aggregated with neighbouring parishes, a final grid resolution of 2 x 2 km might be acceptable. The extent of the MAUP (NOTE: The "P" in MAUP stands for "problem" already if I remember correctly) in the monthly emission maps is not considered to change with season, as the same aggregation zones and distribution methodology are applied for all months.

In this study the initial intention was to calculate emission maps based on both the Unit approach (based on IAEUK) and the Mass Flow approach (based on the NARSES N-flow module with seasonal variability in emission potentials). However, due to the current development stage of the NARSES N-flow module, the linkage between the two models could only be demonstrated in principle rather than fully implemented.

Applying the Unit approach (IAEUK) as input to AENEID is straight forward and requires less data processing than applying outputs from the NARSES N-flow module. However a limitation with the IAEUK-approach is that it fails to incorporate regional variations in the emission potential and that it does not have the flexibility to incorporate different abatement scenarios. Linking the AENEID model to the NARSES N-flow module provides more flexibility as it is possible to calculate spatial emission maps based on various abatement scenarios, to assess scenarios for control/management and also issues such as climate change impacts. However, in addition to further development of the N-flow module, the link between the two systems (AENEID and NARSES) needs to be improved. It is recommended that the N-flow module is improved so that data can be extracted for the whole of the UK in one single operation and in a more streamlined fashion. The NARSES project team should also further develop the N-flow module to incorporate a larger degree of spatial variability in the emission potentials. It is also recommended that the performance of the fine scale modelling (NARSES-AENEID) is be tested against measurement data.

10.7 Potential application of a 1-km dispersion model

The FRAME model currently calculates concentrations and depositions at a 5 x 5 km grid resolution, hence the model fails to reflect the spatial variability at a local scale. Spatial detail is particularly important for NH_3 emissions as, in contrast to many other pollutants, NH_3 deposits at close proximity to the sources, rather than being transported over long distances. Local scale modelling of NH_3 emissions and deposition requires detailed input data (Dragosits *et al.*, 2002; Theobald *et al.*, 2004a), but a field- and farm- based approach cannot be realistically applied for large areas or the whole UK due to lack of detailed input data. Therefore, a new, more pragmatic approach for modelling the dispersion of NH_3 at 1-km grid resolution has been suggested. This 1-km dispersion model would represent a good compromise between fine scale modelling and the crude 5-km approach presently used.

A simple 1-km "dispersion" model has been applied in Switzerland by Rihm (2001) based on the assumption that NH_3 concentrations are proportional to the emission rates and distance from the sources. Based on modelling and measurements, Asman and Jaarsveld (1990) calculated a distance-profile (Figure 10.1) representing the dispersion of NH_3 from a source. Ammonia concentrations decrease with the distance from the source according to Equation 10.1.

$$\text{conc} = 15.197 * \text{distance}^{-1.9332} \qquad \text{(Equation 10.1)}$$

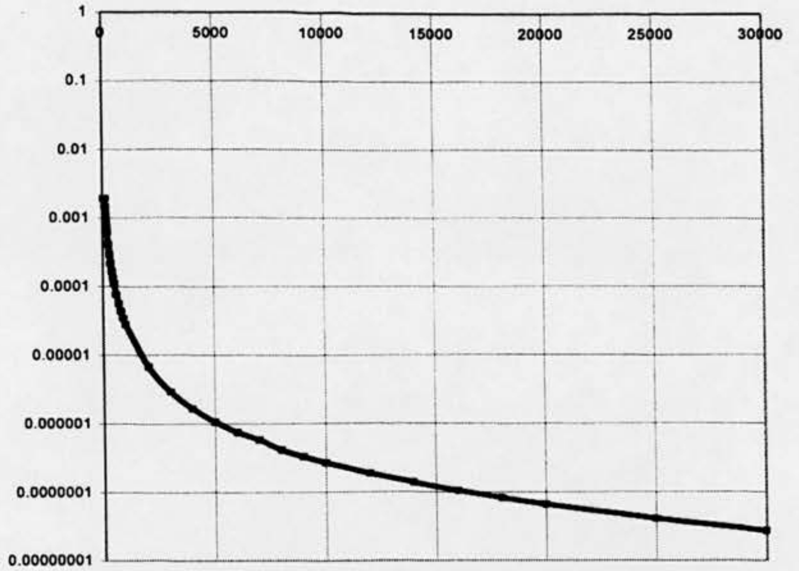

Figure 10.1. Distance-profile representing the typical dispersion pattern of ammonia from a source. The NH_3-concentration (μ/m^3) is a function of the source-receptor distance (m), calculated for a point source situated 3 m above ground, emitting 1 kg NH_3 yr^{-1}. (From Rihm (2001), after Asman *et al.* (1990).

B. Rihm (METEOTEST, pers. comm., 2003) evaluated an approximation of the NH_3-concentration function and applied it to NH_3 sources in Switzerland to calculate the NH_3 concentration in the atmosphere at a 1-km grid resolution (Rihm, 2001). The dispersion model was applied to Switzerland at two levels of resolution depending on the distance from the source:

a) < 4 km : 100 x 100 m resolution

b) 4 - 30 km : 1 x 1 km resolution

Rihm (2001) noted that it was not clear from the distance function (Equation 10.1) what concentration should be applied to the emitting cell where the distance is zero. A zero distance would result in zero concentration from the emitting cell, which is unrealistic as ammonia concentrations are high at close proximity to the source. Rihm (2001) therefore applied the concentration value corresponding to the resolution of the cell, i.e. 100 m, and based this decision on several tests. Emissions at the border of Switzerland were "mirrored" in order to account for deposition from other countries. Rihm (2001) concluded that the resulting dispersion map gave a reasonable picture of the overall NH_3 concentration values in Switzerland. It is therefore suggested that this pragmatic approach should be tested for the UK to further investigate the accuracy of this model, and to compare the results with outputs of the FRAME model. The results of the Swiss model can be aggregated up to 5-km level for comparison with FRAME to evaluate the importance of spatial resolution, i.e. if the 1-km dispersion model can arrive at similar emission results to the FRAME model, despite being based on a much simpler approach.

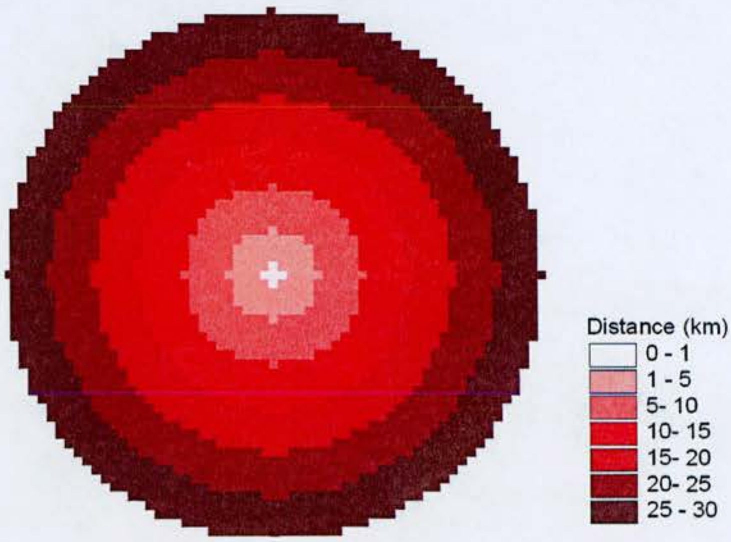

Figure 10.2. Schematic output of a 30 km radius used as a maximum dispersion area for emissions from a source at the centre. The ammonia emission is assumed to decrease with distance from the source, according to function 10.1 and Figure 10.1.

It is suggested that the dispersion model applied in Switzerland should be modified in two ways for application in the UK. Firstly, the emission values should only be "mirrored" at the border to the republic of Ireland, because most of the UK is surrounded by water. Secondly, only one spatial resolution (1 x 1 km) should be used in the modelling process, because the emission map (the AENEID model output) is only available at this resolution. Furthermore, the distance applied for the source cell in the concentration function would need to be tested. Rihm (2001) applied a distance of 100 m (the resolution of the grid cell) for the emitting cell, but applying a 100 m (or even 1 km) distance may not necessarily be the optimal choice when the distance function is applied at 1-km resolution. The 1-km results of the new dispersion model should be aggregated into 5-km resolution to allow for comparison with the FRAME model and to test whether the spatial scale applied in the model, or other factors are more important for the modelling results.

10.8 Other key areas for further work

This study has led to significant improvements of the AENEID model in many key areas. Linking the new AENEID model to the NARSES N-flow module allows the incorporation of regionally varying emission potentials into the AENEID model. Further work is however needed to improve the link between the two models. Furthermore, the NARSES N-flow model needs to be fully developed before outputs from NARSES can be satisfactory applied in the AENEID model. Once this has been achieved, the results of the coupled NARSES-AENEID model should be further tested and validated with measurements.

More work needs to be carried out in developing the temporal aspect of the AENEID model. Further work on the seasonal calculation should concentrate on improving the temporal disaggregation of farming practice, and the incorporation of environmental factors, particularly the effect of temperature. For example, the NAMN measurements show that grazing emissions (shown in sheep dominated areas) are much larger in warmer summer conditions (Sutton *et al.*, 2001c). Ways to incorporate regional variations in temporal aspects should also be further assessed.

This study has showed that the variability of the length of the cattle grazing season significantly influences the cattle emission, both at a regional level within the UK, and between years. Further work should therefore concentrate on improving the modelling of the grazing season and validating output against survey data. Once this has been achieved, the variable cattle grazing season should be fully incorporated into the AENEID model, both at the annual and the monthly level.

The potential to use NH_3 emission maps at a 1-km resolution in atmospheric transport models should be further assessed, as this would improve the accuracy of these models significantly. Before 1-km dispersion modelling can be implemented routinely, the potential conflict with the confidentiality constraints of the disclosive agricultural census data needs to be assessed.

10.9 Conclusions

1. Ammonia emissions are characterised by a high spatial variability at a local scale. When modelling NH_3 emissions, it is important to provide robust emission estimates, since the spatial outputs are used to assess potential environmental impacts, e.g. through the assessment of exceedance of critical loads.
2. The Atmospheric Emissions for National Environmental Impacts Determination (AENEID) model is a tool for spatially dis-aggregating emission inventories. The basic principle of the AENEID model is the spatial distribution of different NH_3 emission stages at the most likely location via a landcover map. The model distributes NH_3 emissions from a range of agricultural activities, such as grazing and housing of livestock, storage and spreading of manures, and fertilizer application, at a 1-km grid resolution over the most likely landcover types.
3. The AENEID model is a FORTRAN model linked with a Geographical Information System. The Fortran model ensures fast and efficient data processing and the GIS-environment is suitable for spatial analysis and modelling.
4. The original AENEID model to spatially distribute ammonia emissions in the UK was developed in the 1990s, and has been improved in several ways in this thesis.
   - The model has been modified to be more flexible to handle spatially and temporally varying emission potentials.
   - The model has been linked to a process-based N-flow module (NARSES), allowing for the incorporation of regionally varying emission source strength and abatement strategies.
   - Monthly emissions have been calculated to estimate the seasonal pattern of ammonia emissions.
   - A sub-model to re-distribute emissions from pigs and poultry more realistically has been developed.
5. The AENEID model was linked to a process based N-flow model (NARSES) in an approach referred to as the coupled NARSES-AENEID model. The results suggest that linking the two models will provide a way to incorporate spatially variable emission potentials and abatement measures, once some limitations in

the output format of the NARSES model are resolved. This model coupling will have the potential to be a powerful system for investigating effects of abatement measures.

6. Seasonal NH_3 emission maps were calculated at a monthly temporal resolution, incorporating seasonal variability both in magnitude and spatial location of emissions. The results show the importance of calculating seasonal estimates, as NH_3 emissions show a strong seasonal emission pattern. Further work is necessary to reduce uncertainties in the monthly emissions, focusing particularly on the spatial dis-aggregation of farming practice and ways to incorporate environmental factors such as temperature.

7. The distribution methodology for pig and poultry emissions from manure spreading had been identified as a major uncertainty in the original AENEID model, as all emissions were assumed to be distributed within the parish of origin. This causes unrealistic emission "hotspots" in the model output, especially in areas with small aggregation zones. In reality manure, particularly poultry manure, is transported significant distances away from the farm. A new iterative distribution methodology to distribute pig and poultry manure was developed in this thesis, which is based on maximum feasible application rates for manure, and also takes account of the manure contribution by other livestock in the area. This approach has been shown to provide more robust emission results.

8. The most important temporal uncertainty in the AENEID emission inventory is the length of the cattle grazing season. Cattle emissions represent the largest agricultural NH_3 emission source in the UK, and because the emission potential from cattle is significantly smaller per unit time during the grazing season, it is important to estimate the length of the grazing season so that the cattle emission potential can be adjusted accordingly.

9. When aggregated, zone-based data, such as the agricultural census data, are used, the results of the analysis depend on the aggregation zone system applied. This phenomenon is referred to as the Modifiable Areal Unit Problem (MAUP). Uncertainties due to the MAUP were investigated by using different aggregation

zone systems for the agricultural census data as input to the AENEID model. The results suggest that the MAUP has a significant effect on the spatial location of emissions.

10. The new AENEID model was applied to dis-aggregate the NH_3 emission inventory in the UK for 2000. The resulting map showed high levels of emissions in areas characterised by intensive agricultural activity such as pig and poultry dominated areas. Cattle emissions are the dominant source in lowland areas, while sheep are the dominant source in hill areas. Emissions from pigs and poultry show a more localised emission pattern. Non-agricultural sources are dominant in urban areas.

11. Changes in NH_3 emissions for the decade between 1990 and 2000 were assessed by applying the AENEID model to census data from 1990, 1996 and 2000, showing that the ammonia emissions have generally decreased during the study period. This decline is mainly explained by declining livestock numbers and changes in farming practice, but also due to less fertilizer application, particularly less urea, which is associated with high emissions of ammonia compared with other mineral fertilizers.

12. The GB NH_3 emission estimate for 1996 based on the new AENEID approach was compared with the original AENEID approach. The two approaches differ in the re-distribution process (the poultry sub-model), the datasets applied in the modelling process (landcover data and agricultural census data), and finally differences in the emission potentials applied. Overall the agricultural emission estimate in GB for year 1996 based on the new AENEID approach was 9 % smaller, mainly due to the new updated emission potentials applied. Furthermore, the new spatial distribution map provides a smoother emission pattern with less extreme emission peaks, due to the new pig and poultry sub-model.

13. Currently, the AENEID model is applied at a 1-km resolution (processing level), but aggregated into a coarser resolution of 5 x 5 km (publication level), to reduce some of the uncertainties in the modelling process, and to ensure confidentiality of the input data. It may however be argued that the data should be made available at the processing level, as this is likely to improve further analysis of

the data, e.g. for application in atmospheric transport models. A disadvantage is, however, that this may conflict with confidentiality constraints in the disclosive input data from the agricultural census data. The basic principles of AENEID ensure that confidentiality of the agricultural census data is maintained by linking emissions to land cover, so that the emission at a 1 km level cannot be directly related to numbers or types of animals from a given farm, as emission outputs are only reported in aggregate form, i.e. livestock emissions and fertilizer/crop emissions rather than cattle, sheep etc.

14. The emission outputs of the AENEID model were validated with measurements, comparing modelled air concentrations of ammonia with monitoring data. The comparison showed a good fit between the model and measurements for the overall magnitude and spatial distribution of NH_3 emissions at a national scale.

15. Further work should concentrate on:
    - Improving the link between the NARSES N-flow module and AENEID. This requires improving the output format of the NARSES N-flow module. When this has been achieved, the results of the coupled NARSES-AENEID model should be further tested and validated with measurements.
    - Improving the temporal disaggregation of farming practice, and also incorporating temporal environmental factors, particularly temperature, in the monthly emission estimate. The possibility to incorporate regional variations in temporal source strength representation should also be further assessed. The results of the monthly AENEID model should be further verified with the FRAME model and independent measurements.
    - The modelling methodology of the cattle grazing season should be further improved, and validated against farm survey data. Furthermore, the modelled grazing season should be incorporated into the AENEID model both at the annual and the monthly level.

16. A pragmatic 1-km grid resolution simple dispersion model tested in Switzerland should be further developed and assessed for UK conditions, and compared with the current atmospheric transport model (FRAME).

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